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Background Document for
Surface Water Screening Procedure

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BACKGROUND DOCUMENT
SURFACE WATER SCREENING PROCEDURE

Prepared by

**Office of Solid Wastes
U. S. Environmental Protection Agency
Washington D.C**

December, 1985

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16. Abstract (Limit: 200 words) The Background Document for the Surface Water Screening Procedure addresses the development of surface water model and how the model was proposed to be used in land disposal restrictions decisions. The potential for exposure of humans and aquatic life to surface water contamination was considered. The surface water analysis encompasses contamination of streams through the interception with ground water contaminated by releases of hazardous constituents in leachate from land disposal units. The surface water screening procedure involves a back calculation using a fate and transport model with a Monte Carlo uncertainty analysis from concentrations that will not adversely affect human health (allowable daily intake) and the environment (Water Quality Criteria) to the constituent concentration in leachate emanating from a land disposal unit. The procedure outlined in this document was proposed to establish acceptable concentrations for specific chemical constituents in waste extracts that may be land disposed. The data used to establish the surface water screening levels are included in this document. Where the back-calculated surface water concentration level for a constituent was more stringent than the screening level developed through the ground water component, the surface water screening level would replace the ground water screening level for purposes of establishing the proposed section 3004(m) treatment standard for that constituent.							
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INTRODUCTION

The Resource Conservation and Recovery Act (RCRA), as amended by the Hazardous and Solid Waste Amendments of 1984, restricts the land disposal of hazardous wastes unless EPA makes a positive finding that continued land disposal of hazardous wastes is protective of human health and the environment, or unless wastes comply with BDAT treatment standards ensuring that threats to human health and the environment are minimized. Land disposal of hazardous wastes frequently leads to release of hazardous constituents to the environment through various media. EPA's Office of Solid Waste (OSW) addresses ground water, surface water, and air contamination of hazardous waste from land disposal sites through the use of mathematical models to determine the screening level concentrations below which land disposal of hazardous constituents may be deemed protective of human health and the environment. Congress has directed EPA to consider:

- A. the long-term uncertainties associated with land disposal;
- B. the goal of managing hazardous wastes in an appropriate manner; and
- C. the persistence, toxicity, mobility, and propensity of hazardous wastes and their constituents to bioaccumulate.

EPA's approach is designed to establish acceptable concentrations for specific chemical constituents in waste extracts by working backward from a point of potential human exposure (through air, ground water, and surface water) to the land disposal unit. The health assessment levels (reference dose) for each constituent

at the point of potential exposure are appropriately partitioned into the air and water to account for exposure through both media. In the approach, decisions to restrict the land disposal of hazardous wastes are being made on a national basis. The RCRA reauthorization legislation establishes a petition process for consideration of site specific factors.

Surface water contamination may result in exposure of humans and aquatic life to hazardous constituents (Figure 1). This potential for exposure necessitated the development of a surface water component for the determination of screening levels. The surface water screening procedure presented in this paper involves the application of an analytic solute transport equation using values for hydrogeologic parameters that are applied in a Monte Carlo routine to derive a distribution of outputs. As illustrated in Figure 2, this analysis encompasses contamination of surface water through interception with contaminated ground water as well as contaminated surface runoff.

Development of the surface water component was guided by the following requirements:

- o the theoretical basis for the approach must be consistent with the approach for the ground water component
- o the surface water component must calculate screening levels for individual constituents using a "back calculation" procedure
- o restriction of wastes from land disposal must be made on a generic basis, not on a site by site basis
- o the approach developed must be expressed as concentrations to permit direct comparisons with a measured leachate value

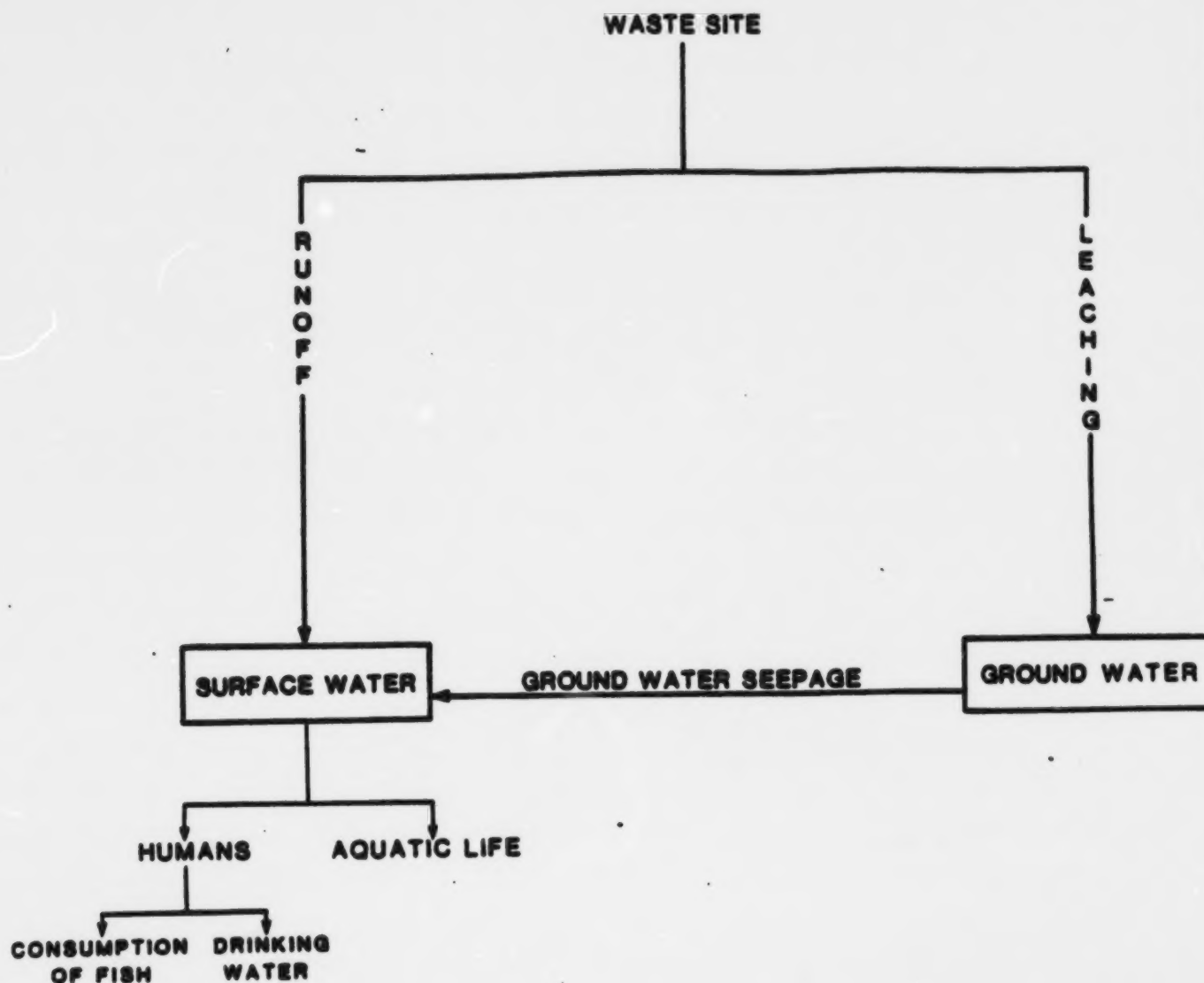


FIGURE 1 —ROUTE OF EXPOSURE FROM LAND DISPOSAL THROUGH THE SURFACE WATER MEDIA

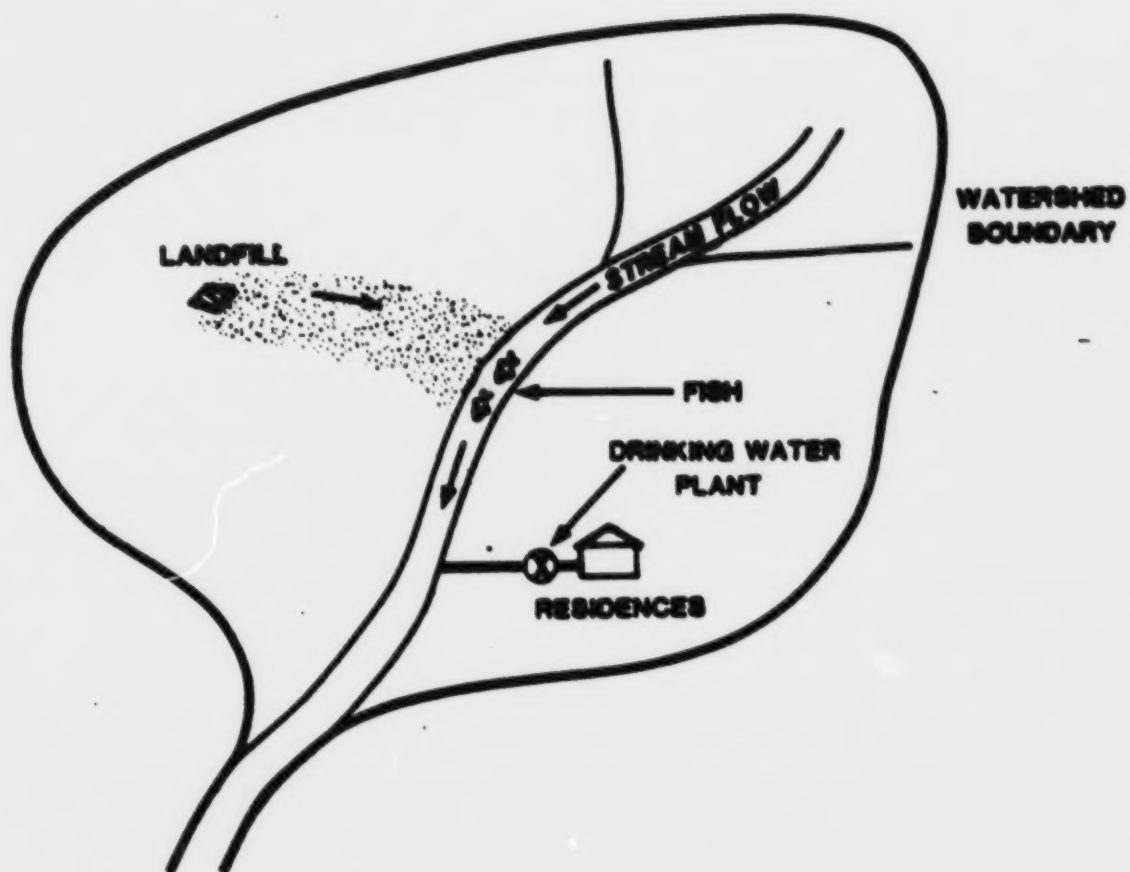


**FIGURE 2 —SCHEMATIC DESCRIPTION OF SURFACE
CONTAMINATION PATHWAYS**

- o the evaluation of organic constituents must be for the parent compound; transformation products are not considered.

The attached documents present information on the development of the surface water screening component that is used to assess hazardous waste for the land disposal restrictions determinations.

A METHODOLOGY FOR ASSESSING SURFACE WATER CONTAMINATION
DUE TO LAND DISPOSAL

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1. INTRODUCTION AND OBJECTIVE

The Hazardous and Solid Waste Amendments of 1984 restrict the continued disposal of hazardous wastes in land-based units at facilities regulated under Subtitle C of RCRA unless EPA makes a positive finding that continued land disposal of hazardous wastes is protective of human health and the environment, or unless wastes comply with BDAT treatment standards established by EPA under Section 3004(m) ensuring that threats to human health and the environment are minimized. Land disposal of hazardous wastes frequently leads to release of hazardous constituents to the environment through various media. As illustrated in Figure 1, this analysis encompasses contamination of surface water through interception with contaminated ground water as well as contamination surface runoff. Surface water contamination may result in exposure of humans and aquatic life to hazardous constituents (Figure 2). This potential for exposure necessitated the development of a surface water component for the development of screening levels. The Agency's technical approach was developed with the assistance of the Environmental Research Laboratory in Athens, Georgia. The theoretical basis for the approach is consistent with the approach for the ground water component.

Development of the surface water component was guided by the following requirements:

- the surface water component must calculate screening levels for individual constituents using a "back calculation" procedure
- restriction of wastes from land disposal must be made on a generic basis, not on a site by site basis
- the approach developed must be expressed as concentrations to permit direct comparisons with a measured leachate value

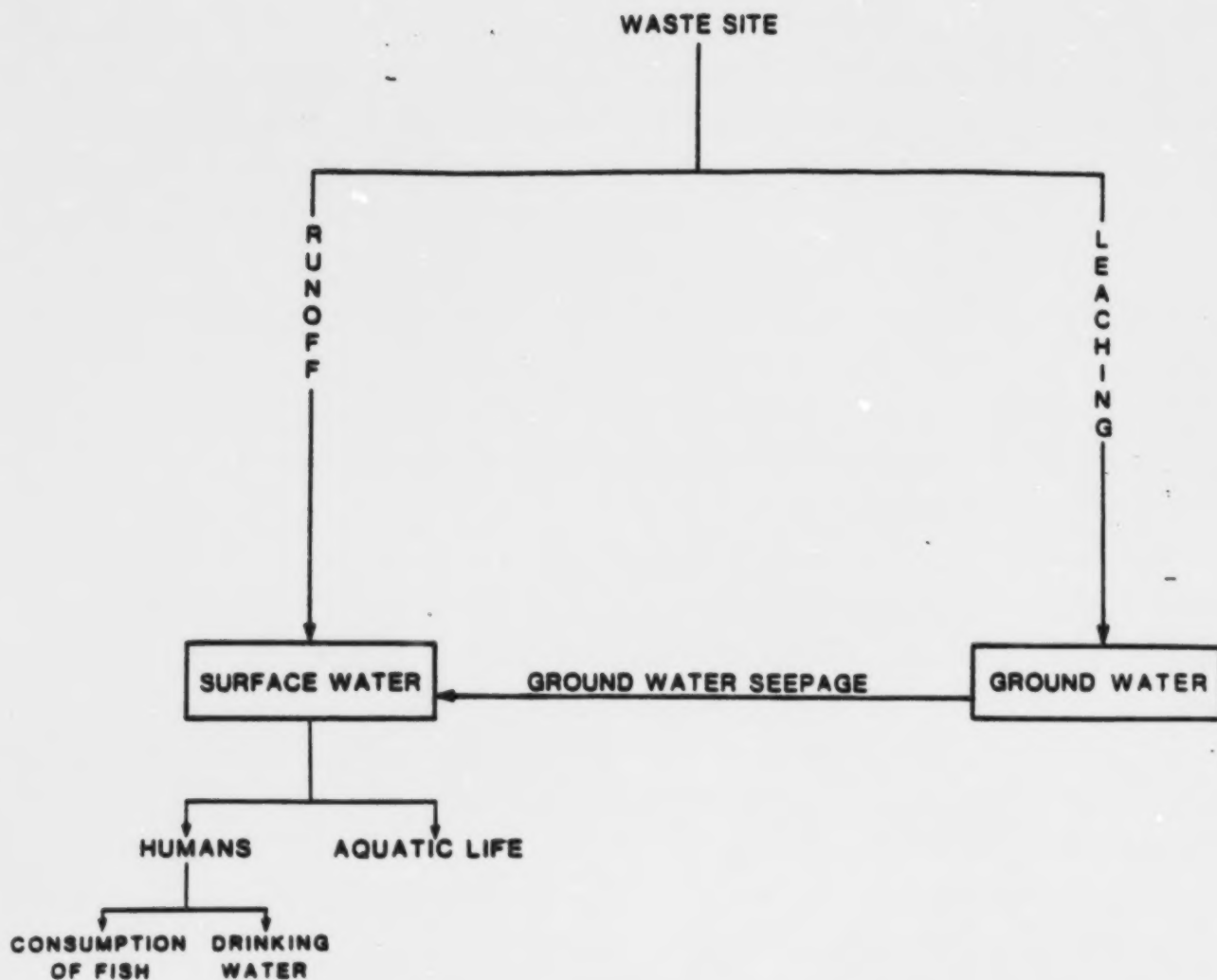


Figure 1
ENVIRONMENTAL SOURCES OF EXPOSURE AND ROUTES OF
EXPOSURE FROM THE SURFACE WATER MEDIA

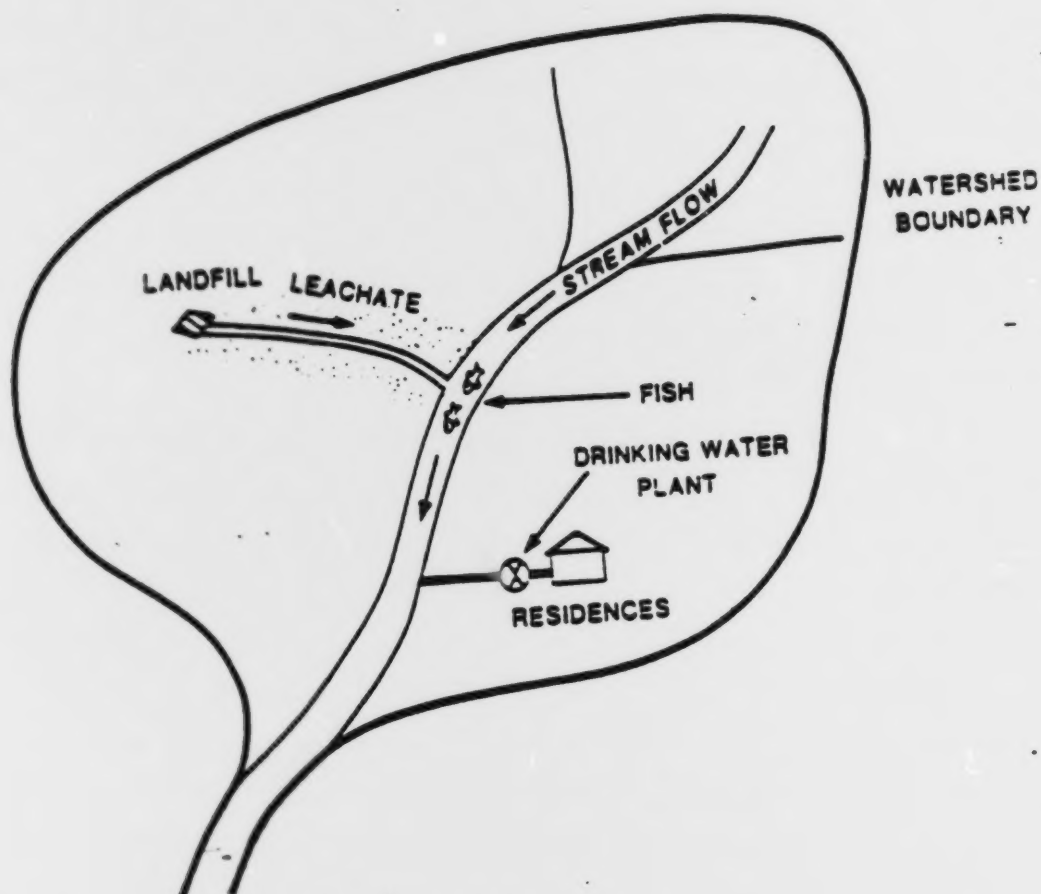


Figure 2
SCHEMATIC DESCRIPTION OF SURFACE WATER
CONTAMINATION PATHWAYS

- o the evaluation of organic constituents must be for the parent chemical; transformation products are not considered.

The methodology assesses surface water contamination resulting from "failure" of the design and operating requirements for land disposal facilities promulgated under Parts 264 and 265 of RCRA. These requirements include: liners, leachate collection and removal systems, ground water monitoring corrective actions, and run-on and runoff controls. This implies both a ground water and a runoff route for chemical release to surface water, as illustrated in Figure 1. Ground water release to surface water occurs as a result of interception of the plume analyzed in the ground water component with the surface water. Direct surface runoff occurs from the "failed" runoff-run-on containment facility.

Three possibilities for surface water contamination are investigated: (1) leaching and subsequent delivery of contaminated ground water to streams; (2) direct runoff from a once in 25-year, 24-hour storm event averaged over a 24-hour period; and (3) direct runoff (catastrophic pulse) from a once in 25-year, 24-hour storm event that is stored for a 24-hour period and then released over a short time period, typically a few hours. The first possibility may occur during or after the lifetime of the facility. The second and third possibility may occur during the lifetime of the facility, before it is capped. The three possibilities for surface water contamination produce two basic routes for human exposure: contamination of drinking water, and contamination of food fish exposed to the ambient stream concentration. Surface water contamination can also cause toxicity in aquatic organisms.

The objectives of the following sections are to describe and implement the procedures developed for evaluating the influence of land disposal on both health-based and environmental impacts. Recall that the overall approach is based on a "back-calculation" to acceptable leachate concentrations given health-based or environmental thresholds that are not to be exceeded at specified exposure points (or routes).

Meeting these overall objectives requires that the following subobjectives or tasks are met:

- o characterize the pathways leading to human and environmental exposure
- o evaluate the likelihood of exposure for each pathway
- o perform "back-calculations" for only those pathways and exposure routes that can result in significant human or environmental exposures
- o determine conditions (chemical- and environmental-specific situations) that will result in allowable leachate concentrations that are more restrictive than for the direct ground water exposure route.

Completion of the above tasks will "screen" the listed chemicals, determine those chemicals that should be restricted based on surface water contamination, and produce maximum allowable leachate levels for appropriate constituents.

2. PATHWAYS LEADING TO CONTAMINATION OF SURFACE WATER AND EXPOSURE TO FISH AND HUMANS

Leachate from land disposal units may enter a stream via ground water intercepting the stream or via surface runoff into the stream. Exposure of humans to chemicals occurs through consumption of fish from the stream or through consumption of drinking water that has been processed through a water treatment plant located downstream from the area of contaminant discharge (Figure 1).

2.1 CONTAMINANT LEACHING AND TRANSPORT IN GROUND WATER SYSTEM

The release and transport of hazardous constituents through the ground water pathway is described in the ground water component. This analysis assumes that the land disposal unit is hydraulically connected to a stream (Figure 3). Upon failure of liners or leachate collectors at the base of the land disposal unit, leachate enters the aquifer directly below the land disposal unit. Precipitation of metals is assumed to occur at this point, placing upper limits on their dissolved concentrations. Dissolved chemicals are then transported through the aquifer under the combined influences of advection and hydrodynamic dispersion as well as retardation and biochemical degradation for nonconservative species. The contaminants discharge into the surface water at the outlet where the aquifer and the stream intercept.

The average concentration, $\bar{C}_g$, at the ground water outlet or surface water entry area, C_g , and the leachate concentration, C_L , may be related by:

$$\bar{C}_g = \zeta_g C_L \quad (1)$$

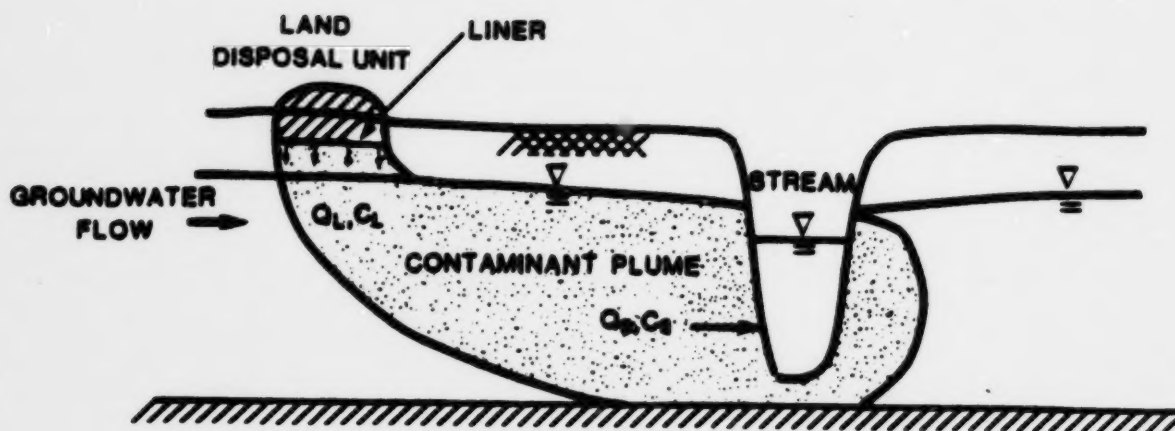


Figure 3

CONTAMINANT TRACKING AND TRANSPORT
IN GROUNDWATER SYSTEM

where the units for contaminant concentration are expressed in mg/l, and ζ_g is a ground water dilution factor accounting for the effects of precipitation, hydrodynamic dispersion, and hydrolysis in the aquifer. In general, ζ_g depends on several subsurface transport parameters. The value of ζ_g is estimated using the analytical transport model developed for the ground water component.

2.2 SURFACE RUNOFF FROM LAND DISPOSAL UNITS

All land disposal systems (landfills, land treatment systems, waste piles, and surface impoundments) must be designed such that at a minimum, runoff from a once-in-25-years, 24-hour storm event is contained (40 CFR Parts 264 and 265). The Agency believes that precipitation events of greater magnitude than the 25-year, 24-hour storm event occur at a sufficiently low probability that the protective design can be considered to provide an acceptable level of performance. This is, by definition, the basis for the performance standard. "Failure" of land disposal systems, illustrated in Figure 4, refers to their inability to contain the 25-year, 24-hour storm event. Direct surface runoff of leachate is assumed to occur from the "failed" containment unit.

For the case shown in Figure 4, the concentration in the runoff from the containment facility is assumed to be the leachate concentration, C_L . This assumption is somewhat conservative: for the case involving surface impoundments, the concentration may be slightly reduced due to dilution when the precipitation water fills the freeboard depth; for the case of landfills and waste piles, not all the precipitation will have contact with the waste.

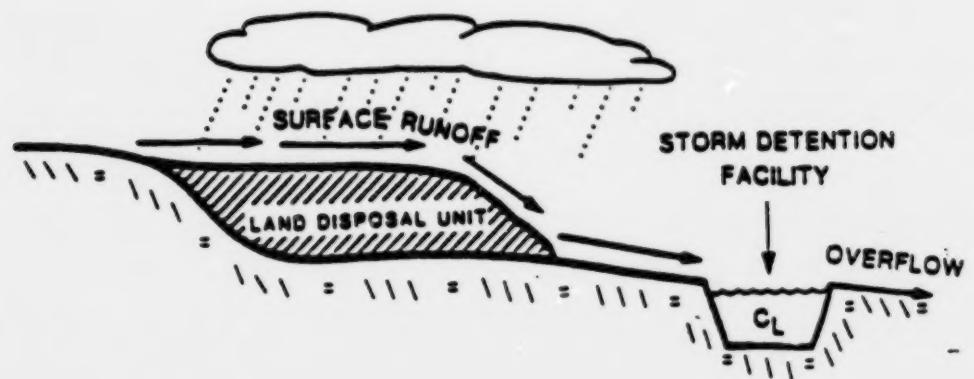


Figure 4
SURFACE RUNOFF FROM LAND DISPOSAL UNITS

Two types of surface runoff are considered. The first assumes containment failure that allows leachate from storm runoff to enter the stream steadily throughout the duration of the 24-hour storm. The second type of surface runoff assumes sudden containment failure that allows leachate generated during an entire storm event to enter the stream as a pulse at the end of the storm. The time over which the steady loading occurs, t_{RS} , is assumed to be equal to 24-hours (the duration of the storm). The time duration of the pulse loading, t_{RP} , is assumed to be between 103 and 104 seconds (approximately 20 minutes to 3 hours). It should be noted that this analysis does not consider particulate flow in the runoff, which is not expected to significantly affect concentrations in the stream.

The concentration of contaminants in the runoff at the area of entry into the stream, C_R , is assumed to be equal to the concentration of the leachate, C_L . This assumption is made in view of the fact that the runoff time is sufficiently small such that the effects of dispersion, adsorption and chemical transformation are negligible.

2.3 LOADING OF CONTAMINANTS TO THE STREAM AND INITIAL DILUTION

As the contaminated water from the ground water or surface runoff enters the stream (Figure 5), it mixes with water supplied by the oncoming streamflow assumed to be free of contaminants. Loading of contaminants to the stream resulting from ground water interception and surface runoff is examined separately. Ground water interception with the stream is assumed to occur continuously. The average concentration of the contaminant in the ground water over the discharge area is equal to $\bar{C}_g$. For the surface runoff release, the contaminant loading is regarded as intermittent

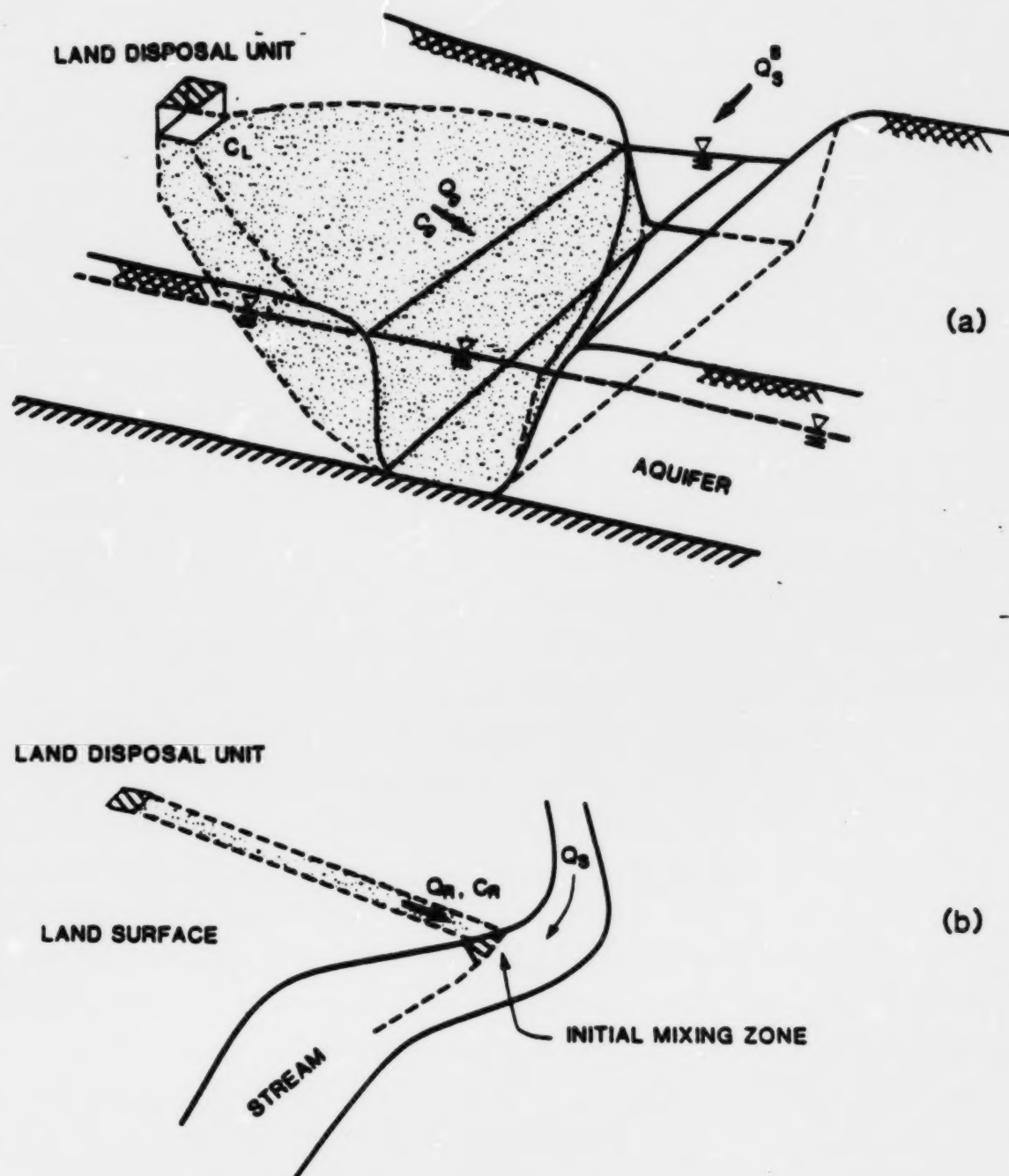


Figure 5

LOADING OF CONTAMINANT INTO THE STREAM, (a) GROUNDWATER LOADING, AND (b) SURFACE RUNOFF LOADING

and occurs over a short time duration, t_{RS} or t_{RP} .

An initial mixing zone in the stream is developed over the contaminant discharge area, A_D . For upland watersheds where the depth of the stream is small, complete, vertical mixing occurs within this mixing zone. However, lateral mixing may be incomplete as indicated in Figure 5b. In the cases of runoff loading, where lateral mixing is likely to be incomplete, there is a finite plume width over which a Gaussian concentration distribution $C_{s,y}$ is assumed. The maximum contaminant concentration and the standard deviation of the Gaussian distribution are denoted by C_R and σ , respectively. The standard deviation, σ , is a measure of the plume width at the edge of the mixing zone.

For ground water loading, the contaminant mass flux being loaded into the stream, $\dot{m}_g$, is the product of the volume of the ground water discharge, Q_g , and the average contaminant concentration in the ground water, $\bar{C}_g$, i.e., $\dot{m}_g = Q_g \cdot \bar{C}_g$. The average contaminant concentration across the mixing zone, $\bar{C}_s$, is related to the average contaminant concentration in the ground water, $\bar{C}_g$, by the following equation:

$$\bar{C}_s = \zeta_s \bar{C}_g \quad (2)$$

where ζ_s is a stream flow dilution factor.

For the two cases of storm runoff and catastrophic pulse loading, the contaminant mass flux being loaded into the stream, $\dot{m}_s$, is the product of the volume of surface runoff, Q_R , and the average contaminant concentration in the surface runoff, C_R , i.e., $\dot{m}_s = Q_R \cdot C_R$. The contaminant concentration distribution across the mixing zone, $C_{s,y}$, is expressed separately for the two types of surface runoff, as follows:

$$C_{s,y} = \zeta_s' C_R \quad (3)$$

and

$$C_{s,y} = \zeta^* C_R \quad (4)$$

where ζ' and ζ^* are stream dilution factors that describe a lateral Gaussian distribution. ζ' and ζ^* are equal to 1 at $y = 0$, and decline to 0 for large values of y .

2.4 TRANSPORT OF CONTAMINANTS DOWNSTREAM

Following initial dilution in the stream, the contaminant is routed downstream from the edge of the initial mixing zone. In the cases of runoff loading, the initial boundary condition assumes a Gaussian distribution $C_{s,y}$. In the case of groundwater loading, the initial boundary condition is the laterally-averaged $\bar{C}_s$. At a specified downstream measurement point, x , (Figure 6) the contaminant concentration, C_x , is expressed as:

$$\bar{C}_x = \zeta_T \bar{C}_s \quad \text{for ground water loading} \quad (5a)$$

$$C_{x,y} = \zeta_T' C_{s,y} \quad \text{for 24-hour storm} \quad (5b)$$

$$C_{x,y} = \zeta_T^* C_{s,y} \quad \text{for pulse runoff loading} \quad (5c)$$

where ζ_T , ζ_T' , and ζ_T^* are reduction factors accounting for the combined influence of advection, longitudinal and lateral dispersion, degradation and sorption occurring during downstream transport. Note that in the case of groundwater loading, lateral mixing is probably almost complete at $x = 0$, due to a fairly extensive contaminant discharge area as compared to the cases of runoff loading.

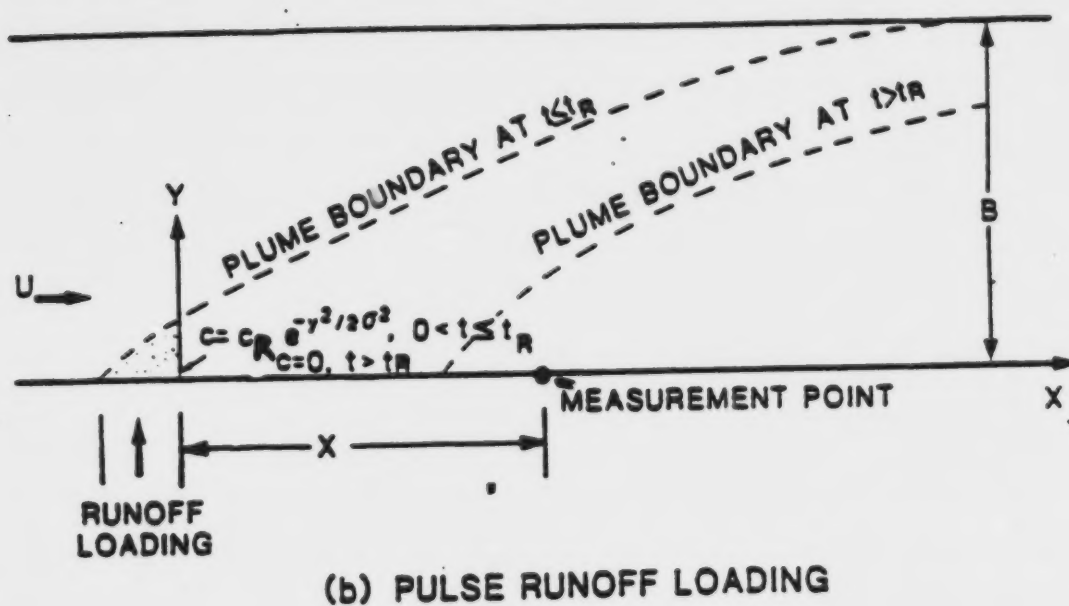
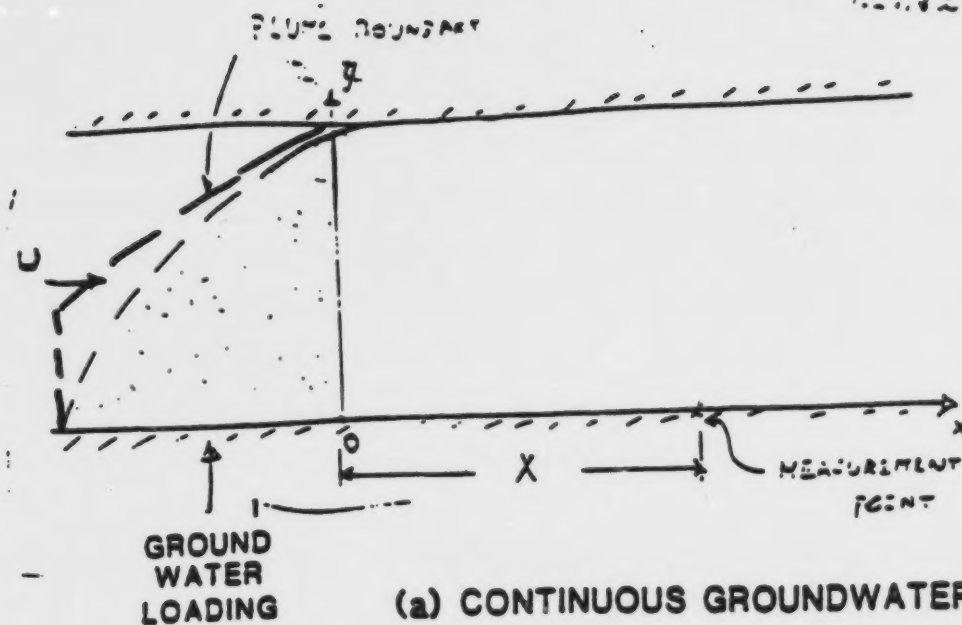


Figure 6
DOWNSTREAM CONTAMINANT TRANSPORT
FROM THE EDGE OF INITIAL MIXING ZONE

2.5 HUMAN EXPOSURE TO CONTAMINANTS THROUGH DRINKING WATER

Humans are exposed to dissolved chemicals through the consumption of water obtained from a treatment plant which is located in the zone of contamination downstream from the initial mixing zone. The plant takes in water from the stream, with a contaminant concentration, $C_{x,y}$. The water is assumed to be treated by a primary settling process allowing suspended solids and adsorbed chemicals to settle out. As a result of this treatment, the contaminant concentration is reduced from $C_{x,y}$ to C_{DW} . The relationship of C_{DW} and $C_{x,y}$ is expressed as:

$$C_{DW} = \zeta_{DW} C_{x,y} \quad (6)$$

where ζ_{DW} is the dilution factor accounting for the reduction in contaminant concentration achieved through the treatment process.

2.6 HUMAN EXPOSURE TO CONTAMINANTS THROUGH CONSUMPTION OF FISH

Another route causing human exposure to chemicals in leachate is the consumption of contaminated fish. To be conservative, it is assumed that these fish reside continuously within the most polluted reach of the stream where concentrations are not reduced by degradation. The allowable daily intake adopted here is based on an average 70 year consumption of contaminated fish.

For the case of runoff loading, the infrequency of runoff events (2 or 3 occurrences in 70 years) and the length of time required for food fish to attain high body burdens (weeks to months) should prevent significant contaminant doses to humans. For the case of groundwater loading, the continuous nature of groundwater seepage into stream may allow fish to attain high body burdens and cause significant contaminant

doses to humans over a life time. Delivery of contaminants through consumption of fish, then, is a long-term average, steady-state process. The analysis will be made allowing for the fact that a fish population can be quite mobile over the length and width of a contaminated stream. Furthermore, a single fish may be exposed to a wide range of concentrations during its life time. The typical fish will be exposed to the average concentration denoted by $\bar{C}_s$. Note that in the case of groundwater loading, $\bar{C}_s$ is given by equation (2).

The chemicals exposed to the fish can be taken up through biochemical exchanges across the gill and gut membranes, and through the skin. When these exchange processes have reached equilibrium, the average concentration in the whole body of the fish becomes C_F , which is related to the exposure concentration by the following equation:

$$C_F = f_D \bar{C}_s K_F = \zeta_F \bar{C}_s \quad (7)$$

where f_D is the dissolved aqueous fraction of the chemical, ($f_D \cdot \bar{C}_s$) is the dissolved aqueous concentration, K_F is a bioconcentration factor depending on both the nature of the chemicals and the species of the fish, and ζ_F is a bioaccumulation factor defined as $\zeta_F = f_D K_F$.

Conceptually, there is one more step in calculating average chemical dose through fish consumption. Most fish are cleaned, with much of the fat removed before consumption. Because organic chemicals are concentrated in fat, another reduction factor could be used to derive fillet concentrations from whole fish concentrations. To be conservative, however, it is assumed that there is no reduction in concentration due to the preparation of fish for consumption.

2.7 DIRECT EXPOSURE OF AQUATIC ORGANISMS

If contaminant concentrations are sufficient, aquatic organisms may suffer chronic toxic effects. Water Quality Criteria have been established by EPA to protect against these effects. These criteria specify acceptable concentrations, durations of averaging periods, and frequency of allowed excursions. To prevent a potential hazard to aquatic life, the average contaminant concentration in the surface water is directly equated to the Criterion Continuous Concentration (CCC) Water Quality Criteria. The duration of the averaging period is set at 4 days, and the frequency of allowed excursions is no more than once in three years. To be conservative, it is assumed that the fish reside continuously within the most polluted reach of the stream where concentrations are not reduced by degradation.

For the case of runoff loading, the infrequency of runoff events (2 or 3 occurrences in 70 years) and their duration (approximately 1 day) should prevent chronic toxic effects in aquatic organisms. For the case of groundwater loading, the continuous nature of groundwater seepage into streams may allow chronic toxic effects. Because groundwater loading is expected to be relatively steady, highest stream concentrations should occur when the stream reaches low, base flow conditions, providing the least dilution water. For these conditions, both loading and dilution are driven by ground water flow. The dilution factor should be steady for a wide range of base flows, as illustrated by Figure 7. The analysis will be made allowing for the fact that a fish population can be mobile over the width of a contaminated stream during a four day period. Thus, the typical fish in the most contaminated stream reach will be exposed to the averaged concentration $\bar{C}_{EXP}$:

$$C_{EXP} = \zeta_{EXP} \cdot \bar{C}_s \quad (8)$$

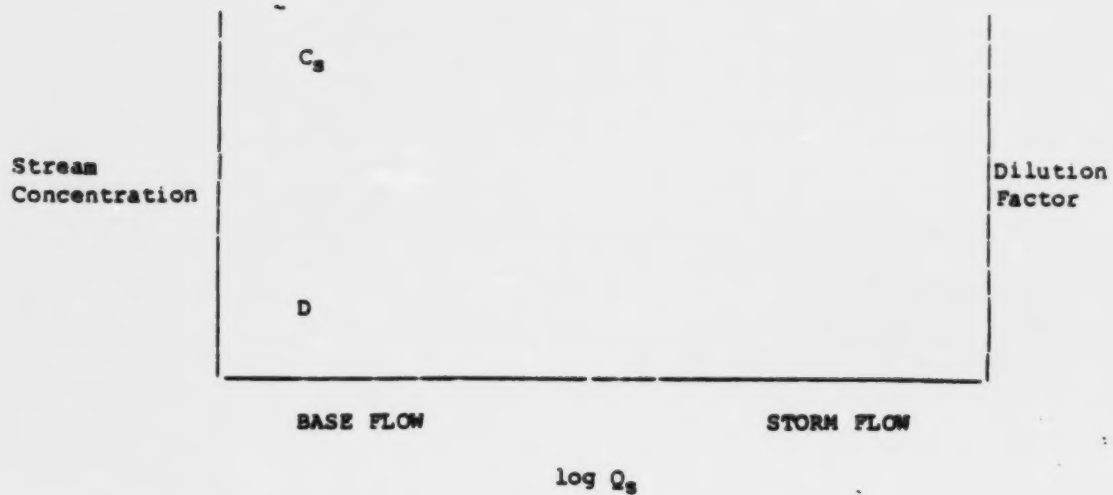


FIGURE 7. VARIATION OF DILUTION FACTOR WITH STREAM FLOW
FOR STEADY GROUNDWATER LOADING

where C_{EXP} is an aquatic exposure factor, here ^{equal} ~~equal~~ to 1.

3. SCENARIOS CONSIDERED

Three sets of scenarios are considered. The three sets (1, 2, and 3) are distinguished from one another by the pathway through which contaminants reach and eventually enter into the stream. Each set consists of three potential exposure routes (A, B, and C) that threaten humans or aquatic organisms.

<u>SCENARIO</u>	<u>ENVIRONMENTAL SOURCES OF EXPOSURE</u>	<u>EFFECT OF THE RELEASE</u>
1A	ground water seepage	human exposure via drinking water
1B	ground water seepage	human exposure via fish consumption
1C	ground water seepage	direct exposure of aquatic organisms
2A	runoff from 25-year 24-hour storm event entering the stream throughout the storm	human exposure via drinking water
2B	runoff from 25-year 24-hour storm event entering the stream throughout the storm	human exposure via fish consumption
2C	runoff from 25-year 24-hour storm event entering the stream throughout the storm	direct exposure of aquatic organisms
3A	runoff from 25-year 24-hour storm event entering the stream as a pulse at the end of the storm	human exposure via drinking water
3B	runoff from 25-year 24-hour storm event entering the stream as a pulse at the end of the storm	human exposure via fish consumption
3C	runoff from 25-year 24-hour storm event entering the stream as a pulse at the end of the storm	direct exposure of aquatic organisms

3.1 SCENARIO 1A: EXPOSURE TO HUMANS THROUGH DRINKING WATER
CONTAMINATED BY LEACHATE CARRIED THROUGH GROUND
WATER TO THE STREAM

This scenario consists of four stages between failure of the waste containment facility and the exposure of the contaminant to humans via drinking water (Figure 8a). Through these stages, the concentration is successively reduced from the leachate concentration, C_L , to the concentration in drinking water, C_{DW} . The relationship between C_{DW} and C_L is given by:

$$C_{DW} = \zeta_g \zeta_s \zeta_T \zeta_{DW} C_L \quad (9)$$

where ζ_g , ζ_s , ζ_T , and ζ_{DW} are attenuation factors due to transport in ground water, mixing at the area of leachate entry into the stream, transport in the stream, and treatment in the drinking water plant.

To determine whether there is a potential health hazard due to surface water contamination, the drinking water concentration can be equated to the specified acceptable daily intake concentration, CADI. Thus:

$$C_{DW} = C_{ADI} \quad (10)$$

and the maximum allowable leachate concentration must be:

$$C_L = \frac{C_{ADI}}{\zeta_g \zeta_s \zeta_T \zeta_{DW}} \quad (11)$$

3.2 SCENARIO 1B: EXPOSURE TO HUMANS THROUGH CONSUMPTION OF FISH CONTAMINATED BY LEACHATE CARRIED THROUGH GROUND WATER TO THE STREAM

This scenario consists of three stages between the containment failure and human exposure via consumption of fish residing in the contaminated surface water (Figure 8b). Through these stages, the input concentration is successively reduced from the leachate concentration, C_L , to the stream concentration, and then increased to the bioconcentrated level in the fish, C_F . The relation between C_L and C_F is given by:

$$C_F = \zeta_g \zeta_s \zeta_F C_L \quad (12)$$

where ζ_F is the bioconcentration factor due to the biochemical exchange processes with the fish.

For back-calculation, the average concentration in the fish, C_F , can be equated to a specified acceptable daily intake bioaccumulation concentration, C'_{ADI} . Thus:

$$C_F = C'_{ADI} \quad (13)$$

and the maximum allowable leachate concentration is given by

$$C_L = \frac{C'_{ADI}}{\zeta_g \zeta_s \zeta_F} \quad (14)$$

3.3 SCENARIO 1C: EXPOSURE OF AQUATIC LIFE DUE TO LEACHATE CARRIED THROUGH GROUND WATER TO THE STREAM

This scenario consists of two stages between the waste containment unit failure and aquatic exposure (Figure 8c). Through these stages, the

input concentration is successively reduced from the leachate concentration, C_L , to the average stream concentration, $\bar{C}_s$. The relation between C_L and $\bar{C}_s$ is given by:

$$\bar{C}_s = \zeta_g \zeta_s C_L \quad (15)$$

For back-calculation, the average concentration in the stream can be equated to a specified Criterion Continuous Concentration Water Quality Criteria, CCC. Thus:

$$\bar{C}_s = CCC \quad (16)$$

and the maximum allowable leachate concentration is given by

$$C_L = \frac{CCC}{\zeta_g \zeta_s} \quad (17)$$

3.4 SCENARIO 2A: EXPOSURE TO HUMANS THROUGH DRINKING WATER CONTAMINATED BY LEACHATE CARRIED THROUGH RUNOFF FROM THE 25-YEAR, 24-HOUR STORM EVENT

This scenario consists of four stages between the waste containment failure and exposure of the contaminant to humans via drinking water (Figure 9a). Through these stages, the concentration is successively reduced from the leachate concentration, C_L , to the concentration in the drinking water, C_{DW} . The relationship between C_{DW} and C_L is given by:

$$C_{DW} = \zeta_s \zeta_T \zeta_{DW} C_L \quad (18)$$

where ζ_s , ζ_T , and ζ_{DW} are dilution factors due to initial mixing at the stream entry area, transport in the stream, and drinking water treatment, respectively.

Because of the pulse runoff loading condition, the concentration C_{DW} is time dependent. Thus, it is averaged over a one-day period. This average concentration, $\langle C_{DW} \rangle$, can then be equated to the specified acceptable daily intake concentration, C_{ADI} . It follows that:

$$\langle C_{DW} \rangle = \langle \zeta_s' \zeta_T' \zeta_{DW} \rangle C_L = C_{ADI} \quad (19)$$

and the maximum allowable leachate concentration is given by:

$$C_L = \frac{C_{ADI}}{\langle \zeta_s' \zeta_T' \zeta_{DW} \rangle} \quad (20)$$

where angular brackets are used to denote the one-day average of the enclosed quantity.

3.5 SCENARIO 2B: EXPOSURE TO HUMANS THROUGH CONSUMPTION OF FISH CONTAMINATED BY LEACHATE CARRIED THROUGH RUNOFF FROM THE 25-YEAR, 24-HOUR STORM EVENT

This scenario consists of four stages between the waste containment failure and exposure of contaminant to humans via consumption of fish (Figure 9b). Through these stages, the input concentration is altered from the leachate concentration, C_L , to the average concentration in the fish, C_F . The infrequency of runoff events (2 or 3 occurrences in 70 years) and the length of time required for food fish to attain high body burdens (weeks to months) should prevent significant contaminant doses to humans from this scenario. Consequently, back-calculation formulas are not developed.

3.6 SCENARIO 2C: EXPOSURE OF AQUATIC LIFE TO LEACHATE CARRIED BY RUNOFF FROM A 25-YEAR, 24-HOUR STORM EVENT

This scenario consists of three stages between the waste containment unit failure and aquatic exposure (Figure 9c). Through these stages, the input concentration is reduced from the leachate concentration, C_L , to the stream concentration $C_{x,y}$. The infrequency of runoff events (2 or 3 occurrences in 70 years) and their duration (approximately 1 day) should prevent chronic toxic effects in aquatic organisms. Consequently, back-calculation formulas are not developed.

3.7 SCENARIO 3A: EXPOSURE TO HUMANS THROUGH DRINKING WATER CONTAMINATED BY LEACHAGE CARRIED THROUGH CATASTROPHIC RUNOFF LOADING TO THE STREAM

This scenario consists of four stages between the waste containment failure and exposure of the contaminant to humans via drinking water (Figure 10a). Through these stages, the concentration is successively reduced from C_L to C_{DW} .

In a similar manner to Scenario 2A, a daily averaged concentration in drinking water, $\langle C_{DW} \rangle$, is obtained and equated to C_{ADI} to yield the following equation for the maximum leachate concentration, C_L :

$$C_L = \frac{C_{ADI}}{\zeta_s \zeta_T \zeta_{DW}} \quad (21)$$

where ζ_s , ζ_T , and ζ_{DW} are, respectively, dilution factors due to initial mixing at the stream entry area, transport in the stream, and drinking water treatment. Note that ζ_s^* and ζ_T^* , instead of ζ_s' and ζ_T' , appear in (21) because of different runoff loading conditions

between Scenarios 2A and 3A.

3.8 SCENARIO 3B: EXPOSURE TO HUMANS THROUGH CONSUMPTION OF FISH
CONTAMINATED BY LEACHATE CARRIED THROUGH
CATASTROPHIC RUNOFF LOADING TO THE STREAM

This scenario consists of four stages between the waste containment failure and exposure of the contaminant to humans via consumption of fish (Figure 10b). Through these stages, the concentration is successively reduced from C_L to C_p . For reasons given in Scenario 2B, significant contaminant doses to humans from this scenario are ruled out. Back-calculation formulas are not developed.

3.9 SCENARIO 3C: EXPOSURE OF AQUATIC LIFE DUE TO LEACHATE CARRIED THROUGH
CATASTROPHIC RUNOFF LOADING TO THE STREAM

This scenario consists of three stages between the waste containment unit failure and aquatic exposure (Figure 10c). Through these stages, the input concentration is reduced from the leachate concentration, C_L , to the stream concentration $C_{x,y}$. The infrequency of runoff events (2 or 3 occurrences in 70 years) and their duration (approximately 1 day) should prevent chronic toxic effects in aquatic organisms. Consequently, back-calculation formulas are not developed.

OVERVIEW OF THE ANALYSES

Scenario 1A: Exposure to humans through drinking water due to ground water interception with the stream

- o Release from unit to ground water
- o Transport in ground water to surface water body

- o Mixing with the stream
- o Transport in stream to drinking water intake
- o Treatment of drinking water

Scenario 1B: Exposure to humans through fish consumption due to ground water interception with the stream

- o Release from unit to ground water
- o Transport in ground water to surface water body
- o Mixing with the stream
- o Uptake by fish through gills, gut, and skin

Scenario 1C: Exposure to aquatic organisms due to ground water interception with the stream

- o Release from unit to ground water
- o Transport in ground water to surface water body
- o Mixing with the stream

Scenario 2A: Exposure to humans through drinking water due to runoff from a 25-year 24-hour storm event entering the stream throughout the storm

- o Surface runoff loading from unit
- o Overland transport assuming no reduction in concentration
- o Mixing with the stream
- o Transport in stream to drinking water intake
- o Treatment of drinking water

Scenario 2B: Exposure to humans through fish consumption due to runoff from a 25-year 24-hour storm event entering the stream throughout the storm

- o Surface runoff loading from unit
- o Overland transport assuming no reduction in concentration
- o Mixing with the stream
- o Transport in stream

- o Uptake by fish through gills

Scenario 2C: Exposure to aquatic organisms due to runoff from a 25-year, 24-hour storm event entering the stream throughout the storm

- o Surface runoff loading from unit
- o Overland transport assuming no reduction in concentration
- o Mixing with the stream
- o Transport in stream

Scenario 3A: Exposure to humans through drinking water due to a 25-year 24-hour storm event entering the stream as a pulse at the end of the storm

- o Surface runoff loading from unit
- o Overland transport assuming no reduction in concentration
- o Mixing with the stream
- o Transport in the stream to drinking water intake
- o Treatment of drinking water

Scenario 3B: Exposure to humans through fish consumption due to a 25-year 24-hour storm event entering the stream as a pulse at the end of the storm

- o Surface runoff loading to the stream
- o Overland transport assuming no reduction in concentration
- o Mixing with the stream
- o Transport in stream
- o Uptake by fish through the gills

Scenario 3C: Exposure to aquatic organisms due to a 25-year, 24-hour storm event entering the stream as a pulse at the end of the storm

- o Surface runoff loading from unit
- o Overland transport assuming no reduction in concentration
- o Mixing with the stream
- o Transport in stream

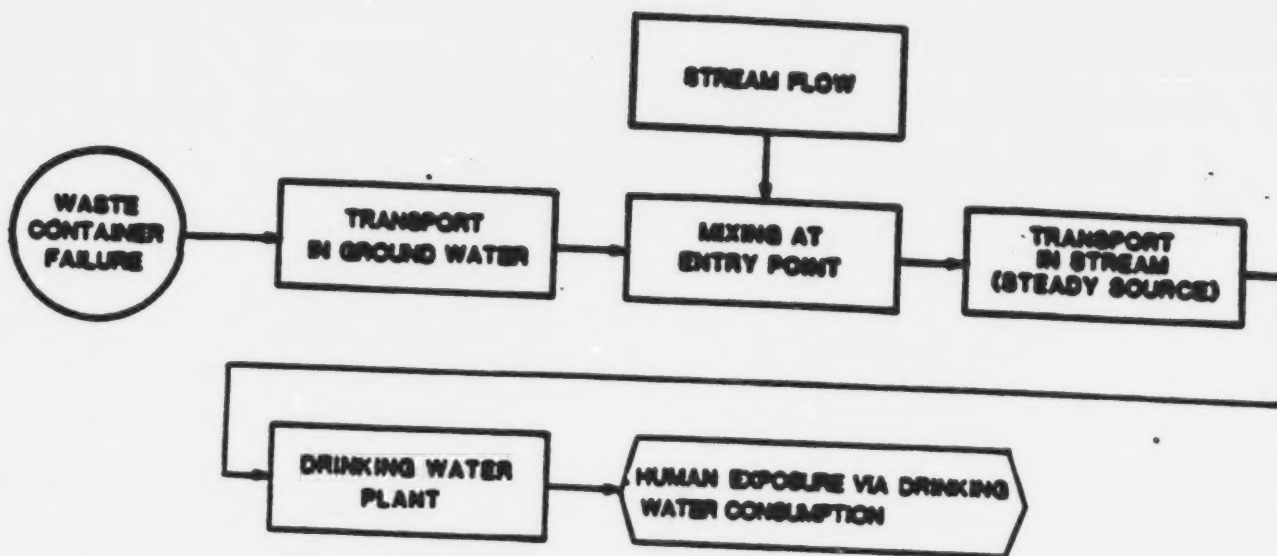


FIGURE 8 (a) --FLOW CHART FOR SCENARIO 1A

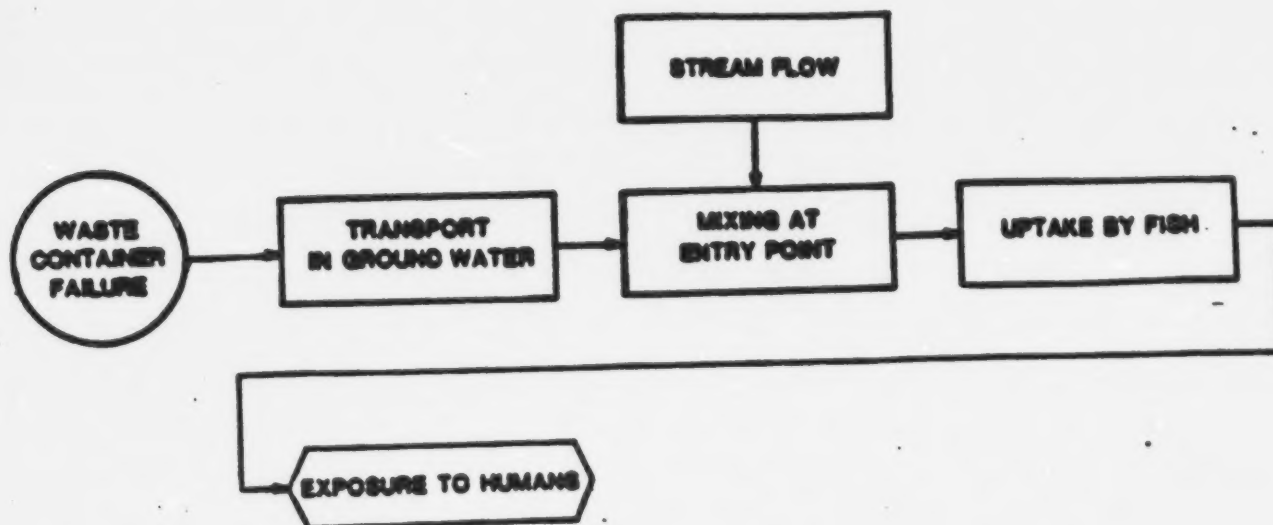


FIGURE 8 (b) -- FLOW-CHART FOR SCENARIO 1B

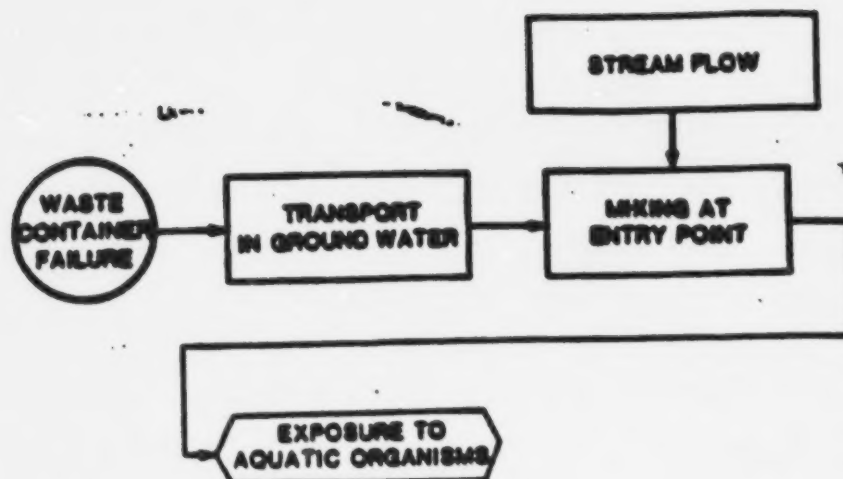
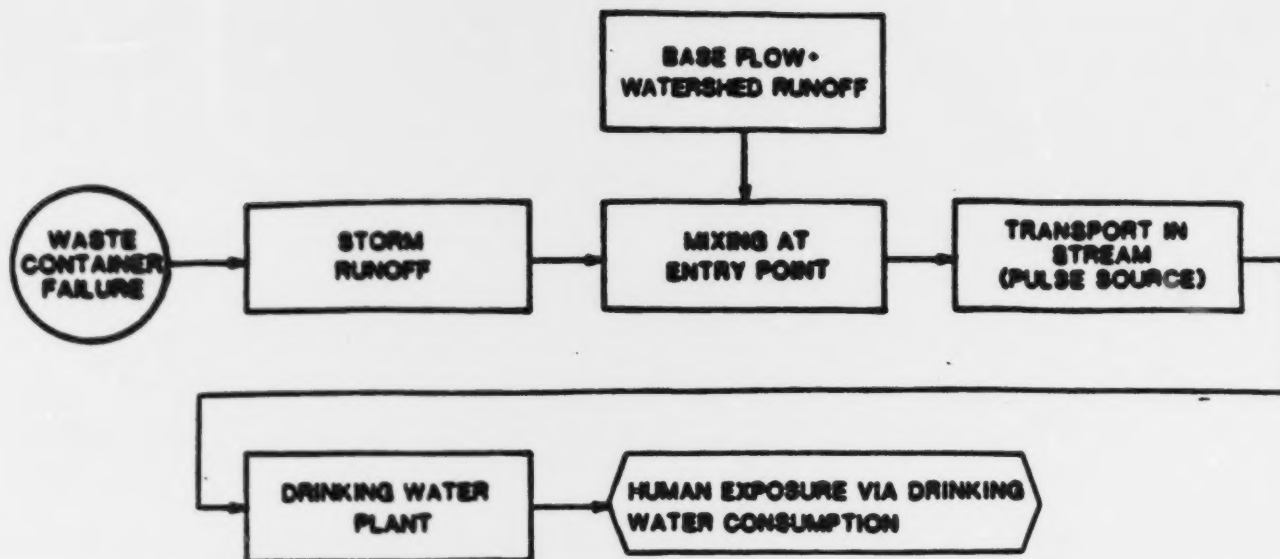
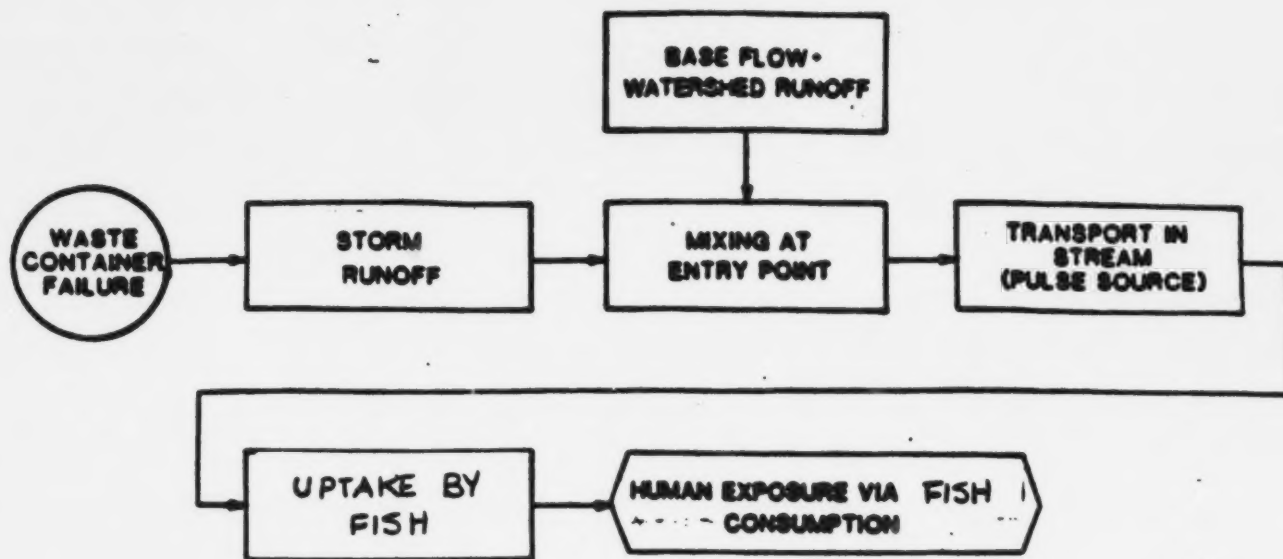


FIGURE 8(c) — FLOW CHART FOR SCENARIO 1C



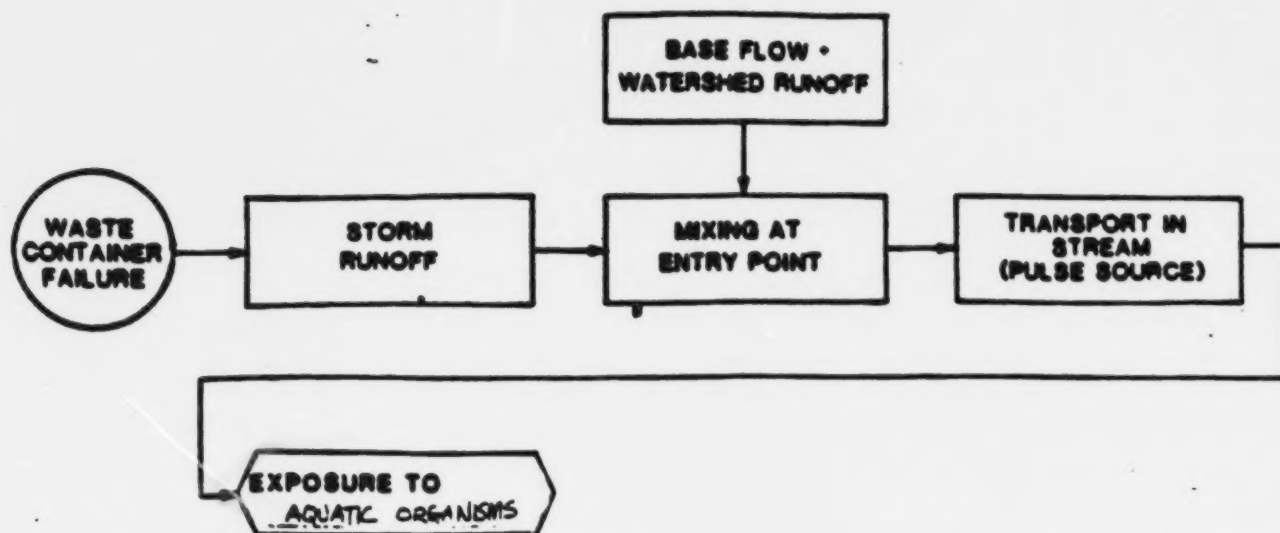
(a) Flow chart for scenario 2A

FIGURE 9a



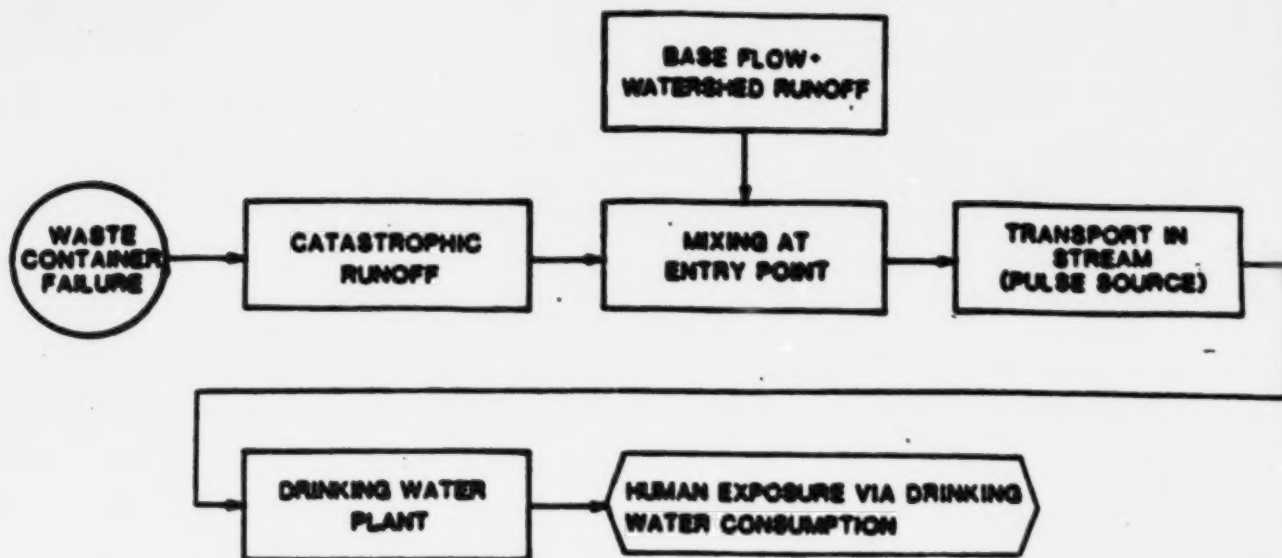
(b) Flow chart for scenario 2B

FIGURE 9B



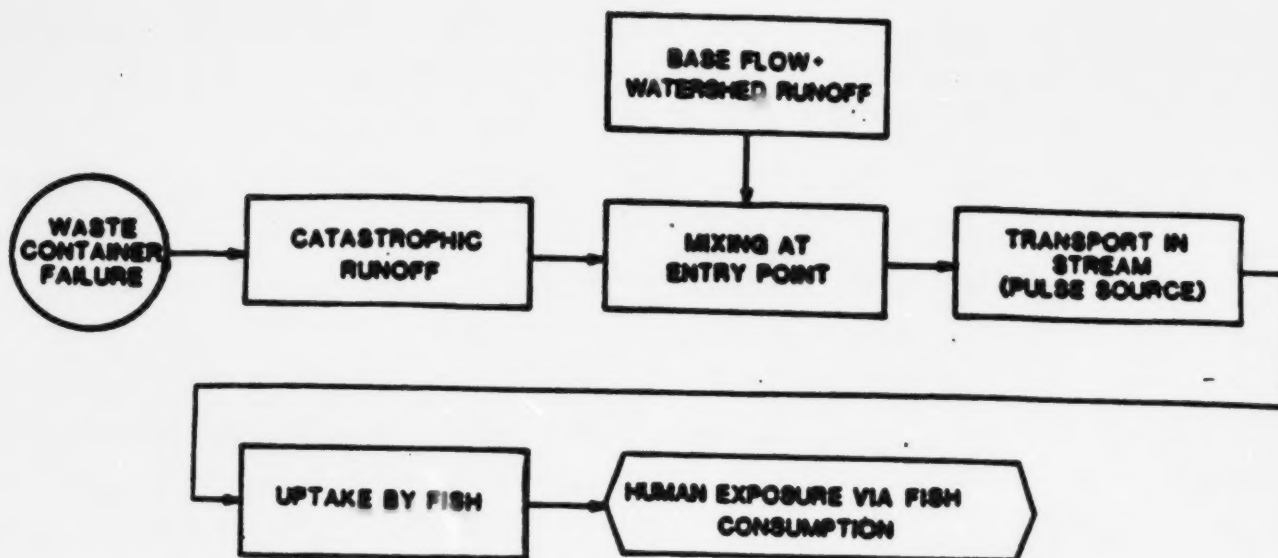
(c) Flow chart for scenario 2C

FIGURE 9C



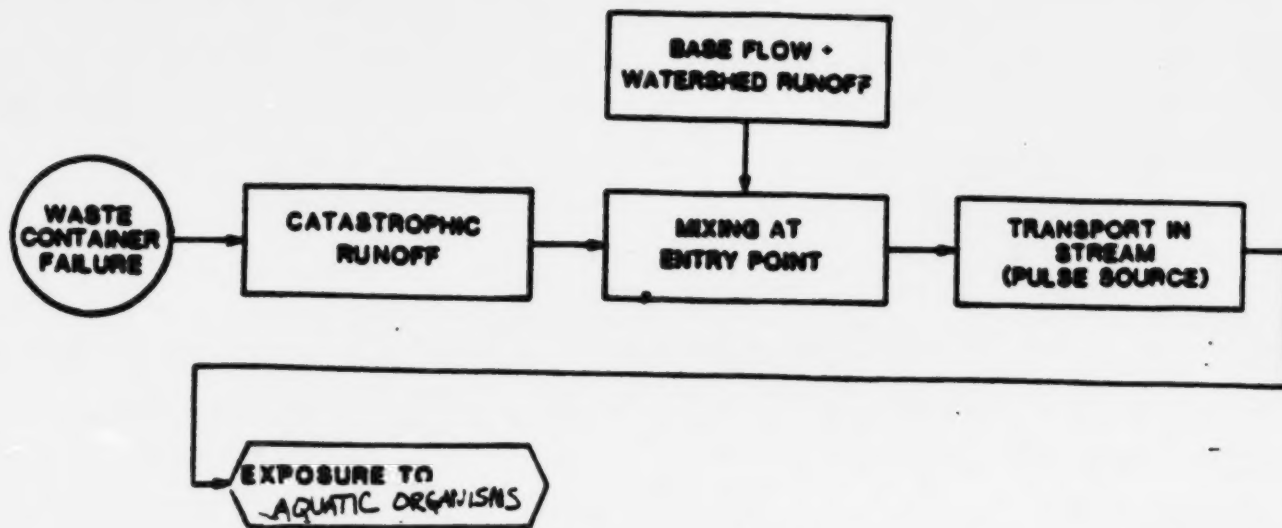
(a) Flow chart for scenario 3A

FIGURE 10a



(b) Flow chart for scenario 3B

FIGURE 10B



(c) Flow chart for scenario 3C

FIGURE 10c

4. EQUATIONS DESCRIBING TRANSPORT AND DILUTION PROCESSES

In this section, equations for determining the dilution factors corresponding to various contaminant pathways are presented. Where appropriate, the key equations are derived. Major symbols used are listed in the notation section.

4.1 LEACHATE LOADING AND DILUTION UPON ENTRY INTO THE STREAM

As mentioned in Section 2, loading of contaminants into the stream can be supplied from ground water flow through an aquifer intercepting the stream, or from surface runoff and overland flow. For the ground water loading, the steady-state loading assumption will be used. The surface runoff loading assumes intermittent pulse loading.

4.1.1 GROUND WATER LOADING AND INITIAL DILUTION

Contaminants reaching a stream via ground water will essentially enter the water body continuously and at the concentration determined from the ground water analysis. Hydraulic impacts of such interception are ignored. That is, the ground water flow field is not influenced by the adjacent surface water. Therefore, for the ground water pathway, the edge-of-stream concentration distribution can be back-calculated to the leachate concentration via the ground water equation, which considers advection, retardation, lateral dispersion, chemical hydrolysis, and initial precipitation (for metals).

For this analysis, it is not necessary to calculate a full three-dimensional concentration distribution for the groundwater. In fact, only the average ground water concentration across the plume is needed to the point of its interception with the stream. The average ground water

reduction factor, which is used in Equation (1), can be calculated using the following mass balance. This mass balance is equivalent to using the three-dimensional ground water equation, then averaging over the width and depth of the plume.

Consider the situation illustrated in Figure 11a. It is assumed that the transport in ground water has reached a steady-state condition. The contaminant mass leaching from the land disposal unit into the ground-water system is given by:

$$\dot{m}_L = Q_L C_L \quad (22)$$

where Q_L is the volumetric rate of percolation through the land disposal and C_L is the leachate concentration. The contaminant plume intercepts the stream over the area, A_g , where the average concentration is denoted by $\bar{C}_g$. Physically, $\bar{C}_g$ corresponds to the average of the actual concentration, which is a Gaussian distribution over the effective flow area A_g . If the groundwater seepage velocity is V_g , then the contaminant mass flux exiting the groundwater system into the stream is given by:

$$\dot{m}_g = V_g A_g \bar{C}_g = Q_g \bar{C}_g \quad (23)$$

At steady-state, the quantities $\dot{m}_g$ and $\dot{m}_L$ are related by:

$$\dot{m}_g = f_H \dot{m}_L \quad (24)$$

where f_H is the fraction of the contaminant mass not transformed by hydrolysis or initial precipitation in groundwater. (Note that Equation (24) is the consequence of mass conservation. Lateral dispersion of the contaminant plume affects the concentration, but not the total mass

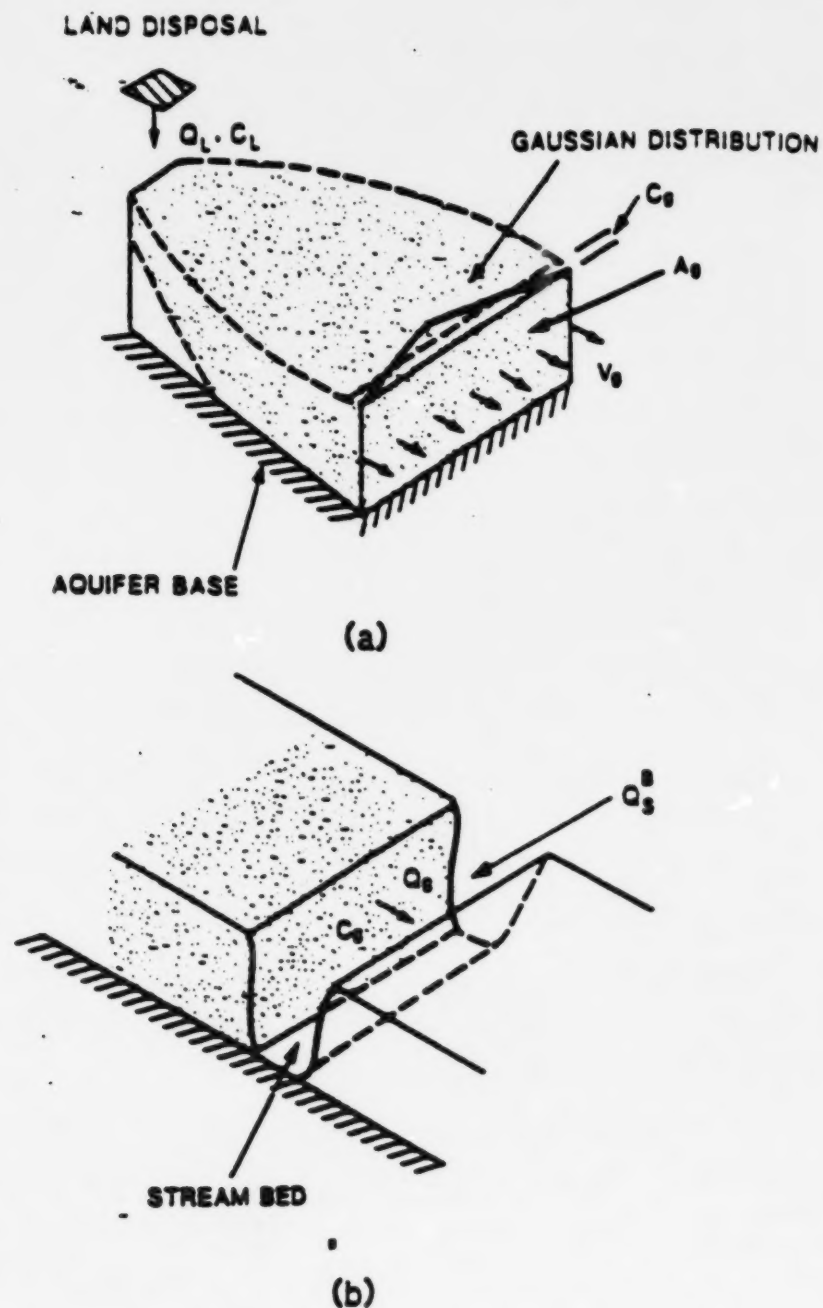


Figure 14

SCHEMATIC DESCRIPTION OF GROUNDWATER CONTAMINANT
LOADING AND DILUTION DUE TO MIXING OF CONTAMINATED
GROUNDWATER WITH FRESHWATER FROM STREAM FLOW

loading to the stream. At steady-state, only chemical transformation reduces the mass loading.) Combining equations (22)-(24), one obtains:

$$Q_g \bar{C}_g = f_H \bar{Q}_L C_L \quad (25)$$

which may be expressed as:

$$\frac{\bar{C}_g}{C_L} = \zeta_g \quad (26)$$

where

$$\zeta_g = \frac{f_H \bar{Q}_L}{Q_g} \quad (27)$$

Equation (27) gives the required expression for the dilution factor due to transport in ground water. The parameters $\bar{Q}_L$ and f_H may be estimated as follows:

$$\bar{Q}_L = \bar{P} (1-f_R) A_w / (86400 \cdot 365.25) \quad (28)$$

and

$$f_H = \exp (-K_g \cdot \tau_g) \quad (29)$$

where $\bar{Q}_L$ = rate of percolation through the land disposal, m^3/sec

$\bar{P}$ = average annual precipitation rate, $m/year$

$1-f_R$ = fraction of precipitation that leaches through waste site to ground water

A_w = surface area of the waste site, m^2

K_g = total effective decay constant in ground water, $years^{-1}$

τ_g = time taken by the contaminant to travel from the land disposal to the stream entry point, years

Values for K_g and r_g can be calculated just as in the ground water analysis. For those chemicals that hydrolyze, K_g is equal to the overall hydrolysis rate constant given by Equations (A23) and (A25) in Appendix A. The travel time of contaminants in ground water is given by

$$r_g = \frac{X_g}{V_g \cdot f_{Dg}} \quad (30)$$

where X_g = distance from site to stream, m

V_g = groundwater seepage velocity, m/yr

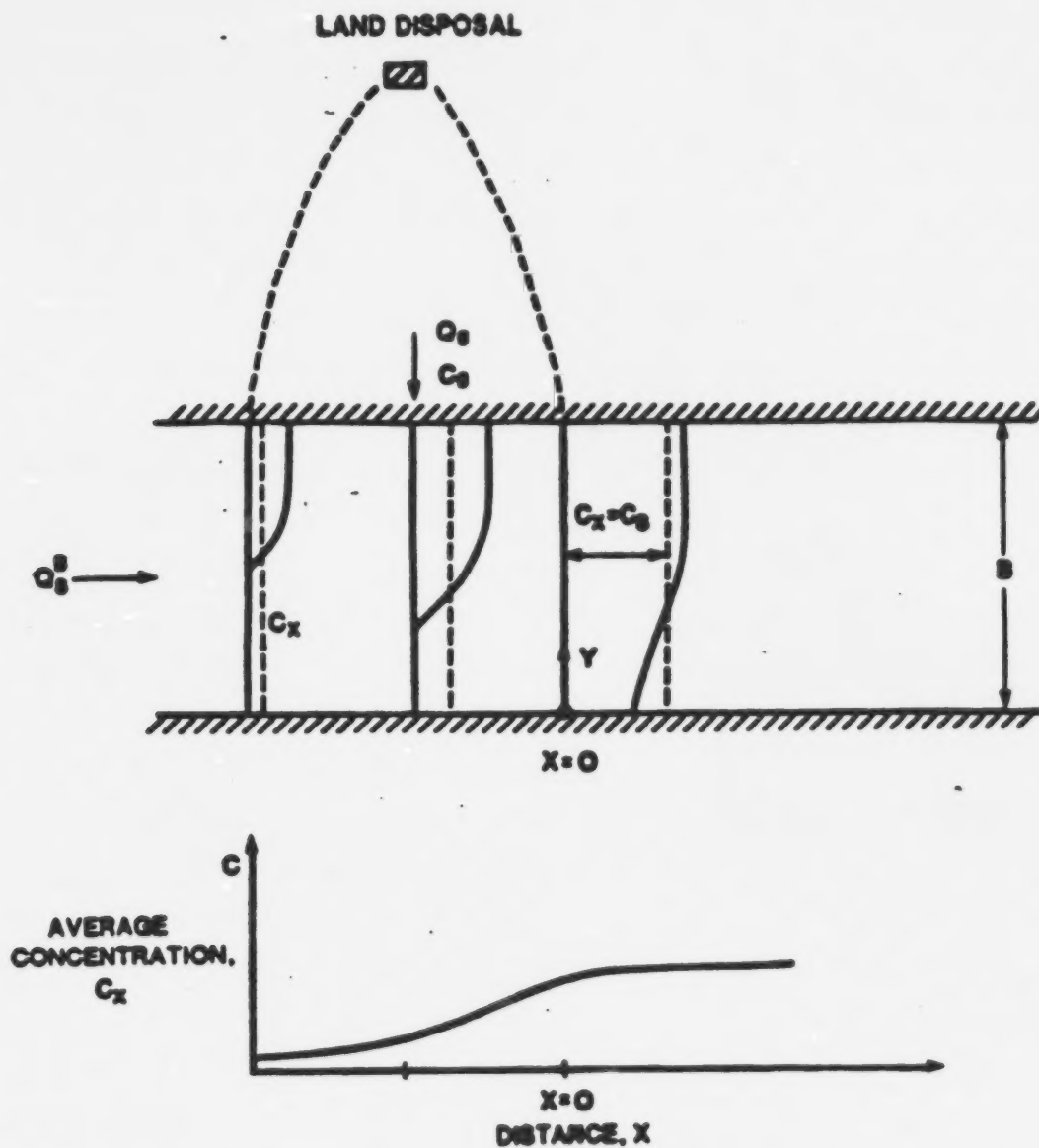
f_{Dg} = fraction of compound that is dissolved (see Equation (A16) in Appendix A)

For metals, f_H represents the fraction not precipitated upon entry to the ground water.

As the contaminated water from the ground water system enters the stream along the side and bottom, it mixes with fresh surface water supplied by the upland watershed as illustrated in Figure 12. Lateral mixing spreads the contaminants until lateral concentration gradients disappear. The laterally-averaged concentration, $\bar{C}_x$, increases with increasing distance reaching a maximum near the downstream edge of the contaminated groundwater plume, where $x = 0$. At the section where $x = 0$, $\bar{C}_x$ corresponds to $\bar{C}_s$ and can be calculated by a simple mass balance. Assuming no in-stream degradation, one obtains

$$Q_s^B \bar{C}_s = Q_g \bar{C}_g \quad (31)$$

which can be rearranged in the form



- Figure 12

GROUNDWATER LOADING TO THE STREAM SHOWING MASS
BALANCE AND CONCENTRATION PROFILES

$$\bar{C}_s = \frac{Q_g}{Q_s^B} \bar{C}_g = \zeta_s \bar{C}_g \quad (32)$$

where Q_s^B is the stream base flow at $x = 0$, and ζ_s is the initial stream dilution factor, which is simply the ratio of the contaminated ground water flow to the stream base flow. Combining (32) and (26) yields the relationship of $\bar{C}_s$ and C_L :

$$\bar{C}_s = \zeta_s \zeta_g C_L \quad (33)$$

which upon substituting for ζ_s and ζ_g gives

$$\bar{C}_s = \frac{Q_g}{Q_s^B} \cdot \frac{Q_L f_H}{Q_g} \cdot C_L = f_H \cdot \frac{Q_L}{Q_s^B} \cdot C_L \quad (34)$$

Note that Q_L is given by equation (28); Q_s^B is given by

$$Q_s^B = \bar{P} (1-f_R) A_s / (86400 \cdot 365.25) \quad (35)$$

where it is assumed that the average annual precipitation rate, P , and the average runoff coefficient, f_R , are the same for the waste site and the entire watershed. Combining equations (A15), (A16) and (A9) gives

$$\bar{C}_s = f_H \cdot \frac{A_w}{A_s} \cdot C_L \quad (36b)$$

4.1.2 STORM RUNOFF LOADING AND INITIAL DILUTION

Contaminants reaching a stream via storm runoff will enter the stream as a steady load throughout the storm duration (Figure 4). The time over which the contaminant loading occurs, t_R , is assumed to be the one-day duration of the storm. Furthermore, it is assumed that the leachate concentration, C_L , is not significantly diluted through runoff, so that $C_R = \zeta_R C_L = C_L$ where C_R is the concentration in the runoff water and ζ_R is the runoff dilution factor = 1. The runoff discharge, Q_R , may be evaluated from

$$Q_R = A_w f_R P_{25}/86400. \quad (37)$$

where P_{25} is the precipitation for the 25-year recurrence 24-hour duration storm, m , and A_w and f_R are as defined previously.

The contaminant mass loading rate $\dot{m}_R$ through runoff is given by

$$\dot{m}_R = Q_R C_L \quad (38)$$

As mentioned in Section 2.3, an initial mixing zone is developed over the runoff discharge area. Within this mixing zone, dilution of the runoff concentration occurs but is somewhat limited by the magnitude of the runoff flow compared with the stream flow. At the edge of the initial mixing zone ($x = 0$), it is assumed that the transverse concentration distribution is a Gaussian distribution defined as

$$C_{s,y} = C_R e^{-y^2/2\sigma^2} \quad (39)$$

where C_R is the peak concentration corresponding to the runoff concentration, and σ is the standard deviation. The contaminant mass flux across the stream section at $x = 0$ is given by

$$\dot{m}_s = U d \int_0^B C_R e^{-y^2/2\sigma^2} dy \quad (40)$$

which may be integrated to give

$$\dot{m}_s = (\pi/2)^{1/2} U d \sigma C_R \quad (41)$$

where U is the mean stream flow velocity and d is the mean depth.

Setting the stream mass flux $\dot{m}_s$ to the runoff mass loading rate gives

$$Q_R C_R = (\pi/2)^{-1/2} U d \sigma C_R \quad (42)$$

Noting that the stream flow Q_s is the product of the mean depth, velocity, and width ($Q_s = UdB$), equation (42) can be solved for σ :

$$\sigma = (\pi/2)^{-1/2} \frac{Q_R B}{Q_s} = 0.798 \frac{Q_R B}{Q_s} \quad (43)$$

Assuming that the runoff coefficient and the average precipitation rate over the watershed are the same as those for the waste site, the stream flow Q_s may be calculated from

$$Q_s = Q_s^B + A_s f_R P_{25}/86400. \quad (44)$$

As indicated in Figure 13a, the contaminant runoff loading is assumed to occur over the time duration, t_R , which is taken to be the 24-hour storm duration, or 86400 seconds.

4.1.3 CATASTROPHIC RUNOFF LOADING AND INITIAL DILUTION

Contaminants reaching a stream via this catastrophic runoff will enter the stream as a steady load over a short period of time $t_R = t_{RP}$. It is assumed that a typical value of t_{RP} is between 0.01 and 0.1 days. The calculation procedure for the catastrophic runoff can be performed in the same manner as that for the storm runoff. Thus, at the edge of the initial mixing zone ($x = 0$), it is assumed that the transverse concentration distribution is Gaussian, as defined by Equation 39. The required parameters C_R and σ are determined from

$$C_R = \zeta_s' C_L = C_L \quad (45)$$

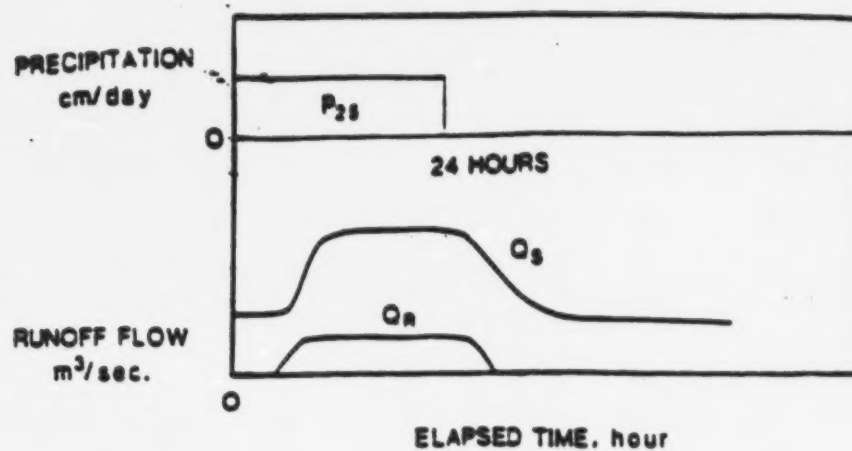
and

$$\sigma = 0.798 \frac{Q_R^*}{Q_s} C_R \quad (46)$$

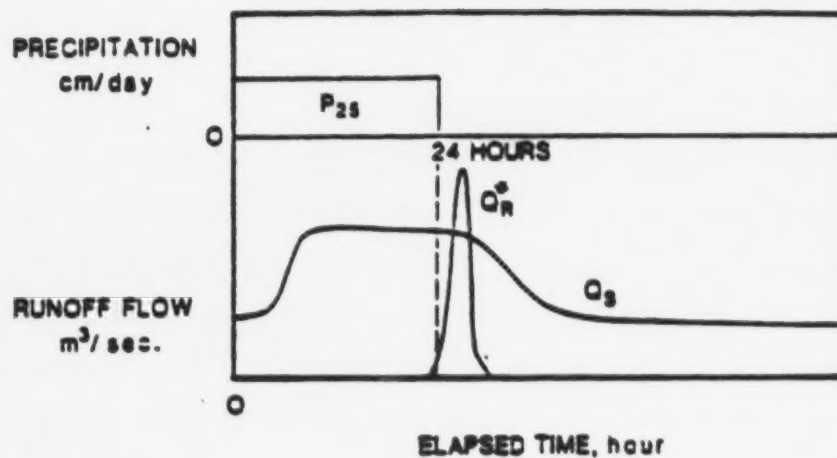
where Q_R^* is the catastrophic runoff discharge (Figure 13b), and is determined from

$$Q_R^* = A_w f_R P_{25}/t_{RP} \quad (47)$$

Note that the units of A_w , P_{25} and t_{RP} are m^2 , m , and seconds, respectively. The stream flow, Q_s , in equation (46) is obtained by summing the base flow, the upstream runoff flow, and the waste site runoff flow. Thus it follows that



(a)



(b)

Figure 13
PRECIPITATION AND RUNOFF FLOWS, (a) RUNOFF DUE TO 24-HOUR
PRECIPITATION, AND (b) CATASTROPHIC PULSE RUNOFF AT THE
END OF THE 24-HOUR STORM

$$Q_s = Q_s^B + (\lambda_s - \lambda_w) f_R f_R^* P_{25}/86400 + \lambda_w f_R P_{25}/t_{RP} \quad (48)$$

where f_R^* is the stream flow recession factor, $0 < f_R^* \leq 1$.

4.2 STREAM TRANSPORT

Following initial dilution in the stream, contaminants at peak concentration, C_s , are routed downstream under the combined influence of advection, longitudinal and lateral dispersion, degradation and sorption. The concentration at downstream distance x from the edge of the initial mixing zone is C_x and is related to C_s via Equation (5), in which ζ_T is the attenuation factor for transport in surface water. The expressions for ζ_T can be obtained from analytical solutions of the transient two-dimensional transport equation. Two cases corresponding to continuous loading and pulse loading of contaminants are considered.

4.2.1 STREAM TRANSPORT BELOW CONTINUOUS GROUND WATER LOADING

The laterally-averaged concentration at the downstream edge of the ground water plume ($x = 0$) is $\bar{C}_s$ (see Figure 12). At a given measurement point located at distance x from the edge of the mixing zone (Figure 6a), the concentration will quickly reach a steady-state value under base flow conditions. The steady-state, laterally-averaged solution for concentrations at the measurement point is given by

$$\zeta_T = e^{-\beta} \quad (49)$$

where $\beta = K.x/U$

K = decay rate constant, sec⁻¹

U = mean downstream velocity, m/sec

(50)

For calculating bioconcentration in Scenario 1b or chronic toxicity in Scenario 1c, we assume that the fish reside continuously in the upstream area where the effect of degradation is insignificant ($x = 0$). Therefore in this case ζ_T is 1, and C_x becomes $\bar{C}_g$.

4.2.2 STREAM TRANSPORT BELOW PULSE RUNOFF LOADING

Pulse runoff loading produces a lateral concentration profile that is a Gaussian distribution characterized (at $x = 0$) by a maximum C_R and a standard deviation of σ over the contaminant loading period t_R (see Figure 6b). Downstream transport will be accompanied by lateral mixing until the contaminants are evenly dispersed across the stream. The concentrations at a given measurement point, x , will increase from zero to a relatively steady value between times (x/U) and $(x/U + t_R)$. At subsequent times, the concentrations will decrease gradually and become zero as the contaminant slug passes through the measurement point. The general analytical solution for transient, two-dimensional transport from a Gaussian pulse source is presented in Appendix B. This solution may be written as

$$C_{x,y,t} = C_g \left[C_f^*(x,y,t) + \sum_{i=1}^{\infty} C_{fi}^*(x,y,t) - C_f^*(x,y,t-t_R) - \sum_{i=1}^{\infty} C_{fi}^*(x,y,t-t_R) \right] \quad (51)$$

where

$$C_f^*(x,y,t) = \frac{\sigma x \exp(U_x/2E_x)}{(2\pi E_x)^{1/2}} \cdot I \quad (52a)$$

For calculating bioconcentration in Scenario 1b or chronic toxicity in Scenario 1c, we assume that the fish reside continuously in the upstream area where the effect of degradation is insignificant ($x = 0$). Therefore in this case ζ_T is 1, and C_x becomes $\bar{C}_s$.

4.2.2 STREAM TRANSPORT BELOW PULSE RUNOFF LOADING

Pulse runoff loading produces a lateral concentration profile that is a Gaussian distribution characterized (at $x = 0$) by a maximum C_R and a standard deviation of σ over the contaminant loading period t_R (see Figure 6b). Downstream transport will be accompanied by lateral mixing until the contaminants are evenly dispersed across the stream. The concentrations at a given measurement point, x , will increase from zero to a relatively steady value between times (x/U) and $(x/U + t_R)$. At subsequent times, the concentrations will decrease gradually and become zero as the contaminant slug passes through the measurement point. The general analytical solution for transient, two-dimensional transport from a Gaussian pulse source is presented in Appendix B. This solution may be written as

$$C_{x,y,t} = C_s [C_f^*(x,y,t) + \sum_{i=1}^{\infty} C_{fi}^*(x,y,t) - C_f^*(x,y,t-t_R) - \sum_{i=1}^{\infty} C_{fi}^*(x,y,t-t_R)] \quad (51)$$

where

$$C_f^*(x,y,t) = \frac{\sigma \exp(-U_x^2/2E_x)}{(2\pi E_x)^{1/2}} \cdot I \quad (52a)$$

$$I = \int_0^t \frac{\exp\left(-\frac{x^2}{4E_x \tau} - \frac{y^2}{4E_y \tau + 2\sigma^2} - \frac{U^2 \tau}{4E_x} - K\tau\right) d\tau}{\tau^{3/2} (2\sigma^2 + 4E_y \tau)^{1/2}} \quad (52b)$$

$$C_{fi}^* = C_f^*(x, 2Bi + 6 \cos(i\pi), t) \quad (52c)$$

E_x = longitudinal dispersion coefficient, m^2/sec

E_y = lateral dispersion coefficient, m^2/sec

The infinite series in equation (51) are evaluated in a computer subroutine using finite sums of N terms, where N corresponds to the number of image sources described in Appendix B. Good convergence is obtained with $N < 10$. The integral I is evaluated numerically using Gaussian quadrature.

4.3 CONTAMINANT DELIVERY TO DRINKING WATER AND FISH

At a distance x downstream, contaminants at concentration C_x may be taken into a drinking water plant or exposed to aquatic organisms, including fish. The drinking water concentration C_{DW} , the aquatic exposure concentration C_{Exp} , and the fish body concentration C_f must be calculated from C_x .

4.3.1 DELIVERY OF CONTAMINANT THROUGH DRINKING WATER

Drinking water plants take in raw water at a distance x downstream from the point of discharge. As a minimum requirement, it is assumed that in any drinking water plant, the raw water having contaminant concentration C_x is treated by allowing suspended solids and adsorbed chemical to settle out. This leads to a reduction of concentration from C_x to C_{DW} . The relationship between C_{DW} and C_x is given by equation (7) with C_{DW}

being the dilution factor corresponding to the fraction of the compound that is dissolved, f_D . The expression for f_D is developed in Appendix A. This may be written as:

$$f_{DW} = f_D = \frac{1}{1 + 0.41 K_{OW} \cdot f_{OC} \cdot S \cdot 10^{-6}} \quad (53)$$

where:

K_{OW} octanol-water partition coefficient, l_{oct}/l_{water}

f_{OC} = organic carbon fraction of sediment

S = sediment concentration, mg/l

4.3.2 DELIVERY OF CONTAMINANTS TO AQUATIC ORGANISMS

Aquatic organisms are exposed to contaminants at a distance x downstream from the point of discharge. Only dissolved species of a compound cross fish membranes and cause internal exposure. There is some evidence, however, that suspended solids with sorbed species can enhance the rate of uptake and thus internal exposure of a compound. The criterion continuous concentration set to protect against chronic toxic effects is generally referenced to the total concentration of a compound. Therefore, C_{EXP} is set to 1 and C_{EXP} is equated to C_X .

4.3.3 DELIVERY OF CONTAMINANTS THROUGH FISH TO HUMANS

Dissolved neutral organic compounds in the water can be taken up by fish through exchange across the gill and gut membranes, and through the skin. Contaminated food can be ingested, resulting in further exchange of compounds across the gut membrane. Concentration levels in the fish will rise until the activity of the compound in the blood equals the

activity of the compound in the water. This condition represents chemical equilibrium. Further uptake of the compound resulting in higher blood concentrations will lead to net exchange out of the fish through the gill, gut, kidney, and skin. Consequently, any chemical buildup above the equilibrium level is controlled by the relative rates of ingestion, metabolism, and exchange. If gill and gut exchange is rapid compared to metabolism, then the equilibrium concentration will not be exceeded, and

$$C_B = f_D C_X \quad (54)$$

where C_B is the dissolved concentration in the blood, mg/l, and f_D is the fraction of chemical dissolved.

If the fish is exposed to steady aqueous concentrations over a long period of time, the distribution of the compound within the various fish tissues will equilibrate, so that:

$$C_l = K_l \cdot C_B \quad (55a)$$

and

$$C_{nl} = K_{nl} \cdot C_B \quad (55b)$$

where:

C_l = lipid (or fat) biomass concentration, mg/kg

K_l = lipid phase partition coefficient, l/kg

C_{nl} = non-lipid (blood, muscle) biomass concentration, mg/kg

K_{nl} = non-lipid partition coefficient, l/kg

The average whole fish concentration C_F (mg/kg) is the weighted sum of the tissue concentrations:

$$C_F = f_L \cdot C_L + (1-f_L) C_{NL} \quad (56)$$

where f_L = fraction of biomass that is lipid. Substituting Equation (54) and (55) into (56) gives:

$$C_F = K_F f_D C_X \quad (57)$$

where K_F is the entire fish partition coefficient, or bioconcentration factor given by:

$$K_F = K_L f_L + K_{NL} (1-f_L) \quad (58)$$

Equation (57) reduces to Equation (8) provided that the parameter ζ_F is defined as:

$$\zeta_F = K_F \cdot f_D \quad (59)$$

Note that unlike the dilution factors, ζ_F is not dimensionless. The unit for ζ_F is l/kg. For strongly hydrophobic compounds, lipid storage dominates K_F . The lipid phase partition coefficient can be replaced by the octanol-water partition coefficient, so that, approximately:

$$K_F = K_{OW} \cdot f_L \quad (60)$$

For less hydrophobic compounds, K_{NL} may contribute significantly to K_F . Non-lipid tissue is composed primarily of water, along with protein and carbohydrates. Assuming that partitioning to non-lipids is always less than or equal to 1% of the partitioning to lipids, a conservative estimate of K_F is approximately (Lassiter, personal communication):

$$K_F = K_{OW} \cdot (f_L + 0.01) \quad (61)$$

For highly polar compounds and metals, the bioconcentration factor K_F can not be estimated from the octanol-water partition coefficient and lipid fraction. In this case, observed field or experimental values of K_F must be used directly.

5. IMPLEMENTATION

Key equations have been presented that together describe leachate loading, dilution, instream transport and transformation, and exposure to humans through drinking water and fish consumption. The important physical parameters can be grouped into those describing the stream and watershed environment, those describing the compound, and those describing the loading/exposure scenario. These are listed in full in the notation section.

The number of potential input variables may appear to make practical application of the methodology described difficult. Fortunately, many variables can be estimated from other more readily available variables.

The proposed methodology has been implemented into a computer code that allows complete analysis of all scenarios described in Section 3.

The computer code can be run interactively in one of two modes. The first mode is called exploration, calculating a permissible CL given specified values for the input variables. The second mode is called Production, calculating a distribution of CL's given distributions for the input variables. Model sensitivity should be investigated in the exploration mode to help guide the effort specifying reasonable input distributions.

* Need example calculations.

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NOTATION

A_g	area through which contaminated groundwater flows into the stream, m^2
A_s	surface area of stream watershed above entry of leachate to stream, m^2
A_w	surface area of water disposal unit, m^2
B	stream width, m
C	concentration of dissolved contaminant, mg/l
C_{ADI}	acceptable daily intake concentration, mg/l
C'_{ADI}	acceptable daily intake fish concentration, mg/kg
CCC	criterion continuous concentration to protect aquatic life, mg/l
C_B	dissolved concentration in fish blood, mg/l
C_{DW}	concentration of dissolved contamination drinking water, mg/l
C_F	average bioaccumulation concentration of dissolved contaminant in fish, mg/kg
$\bar{C}_g$	average concentration across contaminated area of interception between groundwater system and stream, mg/l
C_l	lipid (fat) biomass concentration in fish tissues, mg/kg
C_L	concentration of contaminant in leachate, mg/l
C_{n_l}	non-lipid (blood, muscle) biomass concentration in fish tissues, mg/kg
C_R	concentration of contaminant in the surface runoff, mg/l
$\bar{C}_S$	average concentration of contaminant in the mixing zone adjacent to the stream entry point, mg/l
Λ_{CS}	sorbed concentration, mg/l
C_X	concentration at distance x downstream from the stream entry point, mg/l
d	mean stream depth, m
d_i	impoundment depth, m
d_r	impoundment freeboard required to contain a 24-hr, 25-year storm, m
d_o	mean stream depth at base flow, m

E_x	longitudinal dispersion coefficient, m^2/sec
E_y	lateral dispersion coefficient, m^2/sec
E	longitudinal dispersion factor = $(4E_x (t_r))^{1/2}$
E'	longitudinal dispersion factor = $(4E_x (t-t_r))^{1/2}$
$[E]_i$	environmental property for process "i"
f	depth exponent for stream hydraulic geometry
f_D	fraction of contaminant that is dissolved
f_h	fraction of contaminant mass not transformed by hydrolysis or initial speciation in groundwater
f_l	fraction of fish biomass that is lipid
f_{oc}	organic carbon fraction of suspended sediment
f_{ocg}	organic carbon fraction of aquifer medium
f_R	fraction of precipitation that goes into surface runoff
f_R^*	stream flow recession factor
f_s	fraction of chemical that is sorbed
g	acceleration of gravity, $9.81 m/sec^2$
H	Henry's law constant, $atm \cdot m^3/mole$
$[H^+]$	hydrogen ion concentration = 10^{-PH} mole/l
K	surface water decay rate constant, sec^{-1}
k_{HA}	second-order acid-catalysis hydrolysis rate constant, l/mole-hour
k_{HB}	second-order base-catalysis hydrolysis rate constant, l/mole-hour
k_{HN}	neutral hydrolysis rate constant, $hour^{-1}$
k_i	second-order rate constant for process "i"
K_F	fish partition coefficient (bioaccumulation), l/kg
K_H	Pseudo first order rate constant for hydrolysis, sec^{-1}
K_L	lipid fish tissue partition coefficient, l/kg
K_{nl}	non-lipid fish tissue partition coefficient, l/kg

K_{OC}	organic carbon partition coefficient, l/kg
K_{OW}	octanol-water partition coefficient, l_{oct}/l_{water}
K_{O2}	re-aeration rate constant, sec^{-1}
K_p	equilibrium distribution coefficient for sediment, l/kg
K_v	pseudo first order volatilization rate constant, sec^{-1}
m	velocity exponent for stream hydraulic geometry
$\cdot$	
$\dot{m}_g$	contaminant mass flux exiting the groundwater, g/sec
$\cdot$	
$\dot{m}_L$	contaminant mass flux leaching from land disposal unit, g/sec
$\cdot$	
$\dot{m}_R$	mass loading rate to stream from surface runoff, g/sec
MW	molecular weight of compound
n	Manning roughness coefficient, $sec/m^{1/3}$
$[OH^-]$	hydroxide ion concentration = 10^{-pOH} mole/l
pH	stream pH
pH _g	groundwater pH
pOH	stream pOH
pOH _g	groundwater pOH
P	transformation product, mg/l
$\bar{P}$	average annual precipitation, m
P_{25}	average precipitation for 25-year, 24-hour storm, m
q	average flow per unit area, $m^3/sec/m^2$
Q	stream flow, m^3/sec
Q_g	contaminated groundwater discharge, m^3/sec
Q_L	rate of percolation through the land disposal, m^3/sec
Q_R	storm runoff flow, m^3/sec
Q_R^*	flow rate of spill delivered to the stream, m^3/day
Q_S	stream flow, m^3/sec
Q_S^B	stream base flow, m^3/sec

R	ideal gas constant, $8.206 \times 10^{-5} \text{ m}^3\text{-atm/mol}^\circ\text{K}$
R _G	gas phase resistance, sec/m
R _i	reaction rate for process "i", mg/ l-sec
R _L	liquid phase resistance, sec/m
s	channel slope
S	sediment concentration, mg/l
S'	sediment concentration, kg/l
t	time since beginning of discharge, sec
t _R	time duration of the contaminant loading, sec
T	water temperature, °C
T'	water temperature, °K = 273 + T
U	mean downstream advective velocity, m/sec
U*	shear velocity, m/sec
V _g	groundwater seepage velocity, m/year
W	wind speed at 10 cm above surface, m/sec
WAT	water vapor exchange constant, m/sec
W _z	wind speed at height z, m/sec
x	downstream distance from discharge, m
Y _i	yield coefficient for process "i"
Z	wind measurement height, m
α	acid-catalysis hydrolysis rate enhancement factor for sorbed compound
β	kinetic reduction exponent = $k \cdot x/U$
β ₁	exponent in plane source equation = $xU/2E_x$
β ₂	exponent in Plane source equation = $xw/2E_x$
w	advective-dispersion factor, m/sec
ζ _{EXP}	aquatic exposure factor
ζ _{DW}	drinking water treatment reduction factor

ζ_F	biomagnification factor in fish, l/kg
ζ_g	groundwater attenuation factor
ζ_S	dilution due to mixing at the stream entry Point
ζ_S'	stream flow dilution factor (pulse load for first runoff event)
ζ_S^*	stream flow dilution factor (pulse load for second runoff event)
ζ_T	attenuation factor for transport in the stream (groundwater loading)
ζ_T'	attenuation factor for transport in the stream (surface runoff loading)
ζ_T^*	attenuation factor for transport in the stream (catastrophic runoff loading)
τ_g	time taken by the contaminant to travel from the land disposal unit to the stream entry point

APPENDIX A
ADVECTION, DISPERSION AND CHEMICAL TRANSFORMATION
IN STREAM

In this appendix procedures and formulas for estimating physical parameters of advection, dispersion and chemical transformation in surface water are presented.

1. Advection

A compound introduced to a water body will be advected downstream with the bulk water at mean velocity U such that

$$U = Q / (B \cdot d) \quad (A1)$$

where Q = stream flow, m^3/sec

For a given flow in a specific stream reach, width, depth, and velocity are related empirically by the following equations (Leopold and Maddock, 1953):

$$B = aQ^b \quad (A2a)$$

$$d = cQ^f \quad (A2b)$$

$$U = kQ^m \quad (A2c)$$

where the sum of the exponents ($b+f+m$) and the product of the constants ($a \cdot c \cdot k$) must equal 1.0. Although theoretical considerations predict that $b = 0.23$, $f = 0.42$, and $m = 0.35$, considerable variations have been observed among sites. Figure A.1 presents the exponents observed at 139 sites, as analyzed by Park (1977).

The stream base flow for subwatersheds can be calculated from the relationship where

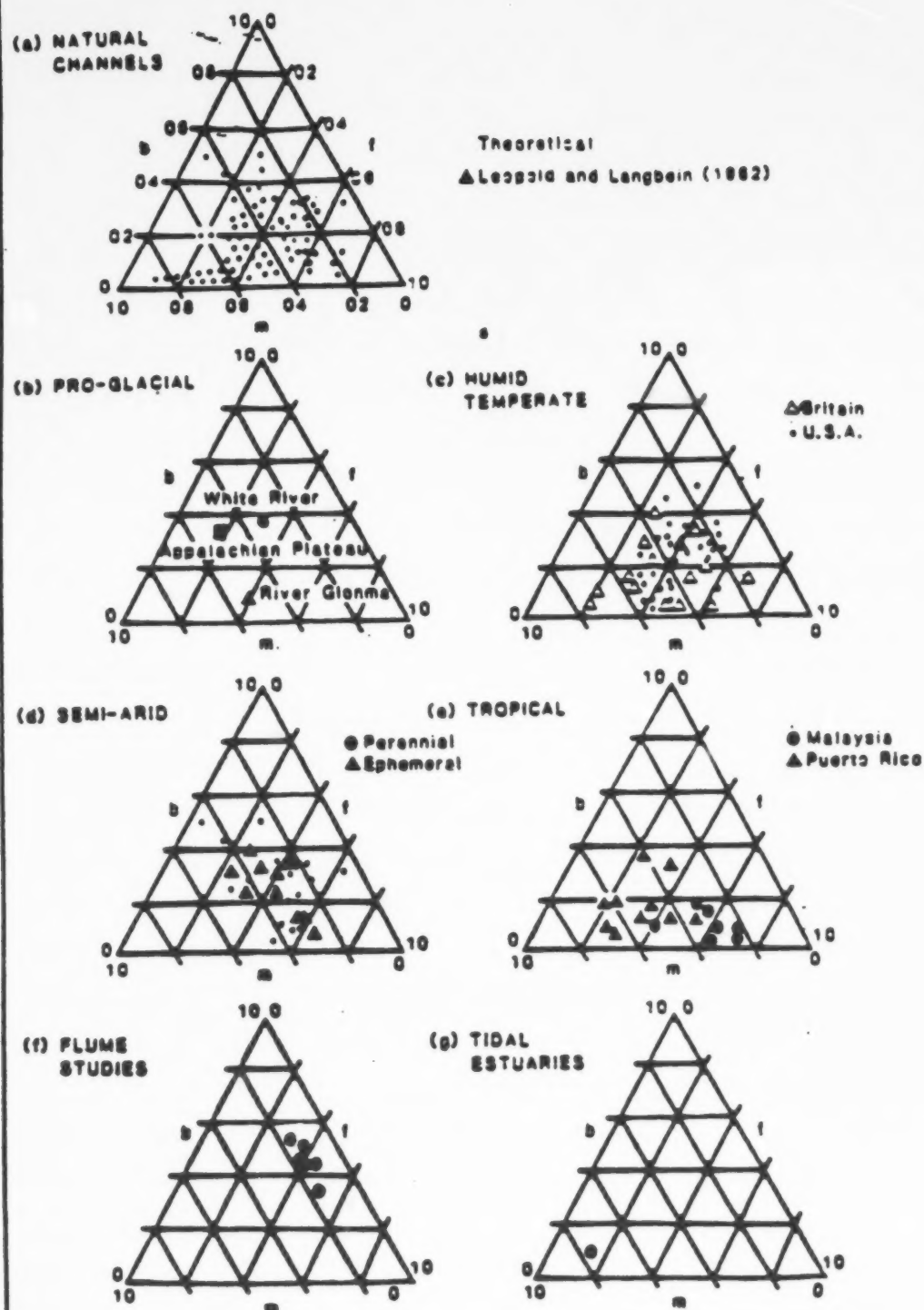


Figure A.1

TRI-AXIAL GRAPHS OF AT-A-STATION
HYDRAULIC GEOMETRY EXPONENTS

$$Q_0 = A_s \cdot q \quad (A3)$$

where q = average flow per unit area $\frac{m^3/sec}{m^2}$

Velocity at baseflow, U_0 can be calculated by Manning's equation: (A4)

$$U_0 = d_0^{2/3} \cdot s^{1/2} / n$$

where d_0 = depth baseflow, m

s = channel slope, m/m

n = Manning's roughness coefficient, sec/m^{1/3}

The width at baseflow B_0 , can be calculated from U_0 , d_0 , and the baseflow Q_0 using equation (A1) rearranged:

$$B_0 = Q_0 / (U_0 \cdot d_0) \quad (A5)$$

The stream flow during a storm includes both baseflow and runoff, as given by operation (39)

$$Q_s = Q_s^B + A_s P_{25} f_R \quad (A6)$$

Given the baseflow values B_0 , d_0 , U_0 , and Q_0 and the stormflow value Q , the widths, depths, and velocities for stormflow conditions can be calculated as

$$B = B_0 (Q/Q_0)^b \quad (A7a)$$

$$d = d_0 (Q/Q_0)^f \quad (A7b)$$

$$U = U_0 (Q/Q_0)^{1-b-f} \quad (A7c)$$

When the theoretical values for b and f hold, U increases with Q to the 1/3 power. A ten-fold increase in flow, then, results in a doubling of velocity.

Streamflows and the associated hydraulic variables, then, can be synthesized from distributions of watershed areas A_s , areal flows q , channel slopes, s , channel roughness factors n , precipitation totals P_{25} , runoff coefficients f_R , and the hydraulic geometry exponents b and f .

2. Dispersion

A compound advected through a water body will be mixed vertically, laterally, and longitudinally from areas of high concentration to areas of low concentration. The rate of mixing is proportional to the concentration gradient and either a turbulent mixing coefficient or a dispersion coefficient. A turbulent mixing coefficient in rivers is proportional to the length scaled and the intensity of turbulence, which is represented by the shear velocity:

$$U^* = \sqrt{g \cdot d \cdot s} \quad (A8)$$

where U^* = shear velocity, m/sec

s = channel slope

d = mean depth, m

g = acceleration of gravity, m/sec²

Because vertical mixing in streams occurs very quickly, we assume completion during the initial dilution stage. Lateral mixing is most important in the near field. It is smallest for uniform straight channels, and increases with curves and irregularities. Fischer et al. (1979) suggest calculating the lateral diffusion coefficient as:

$$E_y = 0.6 \cdot d \cdot U^* \quad , \pm 50\% \quad (A9)$$

The proportionality factor can vary evenly between 0.4 and 0.8.

Longitudinal turbulent mixing is generally much smaller than shear flow dispersion, which is caused by velocity gradients. Fischer et al., suggest calculating the longitudinal dispersion coefficient with the approximate relationship:

$$E_x = 0.011 U^2 \cdot B^2 / d \cdot U^* \quad (A10)$$

Here, again, the Proportionality factor can vary $\pm 50\%$.

3. Chemical Transformation

A compound transported through a water body can undergo several physical and chemical transformations. Fast reactions are treated by assuming local equilibrium conditions. Sorption is considered to be in with desorption:

$$S' + C_w = C_s \quad (A11)$$

Where S' = sediment concentration, kg/l

C_w = dissolved aqueous concentration, mg/l

C_s = sorbed concentration, mg/l

The local equilibrium concentrations C_w and C_s are governed by the equilibrium distribution coefficient K_p (l/kg):

$$K_p = \frac{C_s}{S' \cdot C_w} \quad (A12)$$

It has been shown that for sorption of hydrophobic organic compounds:

$$K_p = K_{oc} \cdot f_{oc} \quad (A13)$$

where K_{oc} = organic carbon partition coefficient, l/kg

f_{oc} = organic carbon fraction of sediment

and K_{OC} has been correlated with K_{OW} (Karickhoff et al., 1979):

$$K_{OC} = 0.41 \cdot K_{OW} \quad (A14)$$

Combining Equations (A12) - (A14) and rearranging terms gives an expression for the fraction of the compound that is dissolved:

$$f_D = \frac{1}{1 + 0.41 \cdot K_{OW} \cdot f_{OC} \cdot S \cdot 10^{-6}} \quad (A15)$$

The sorbed chemical fraction f_s is equal to $1 - f_D$.

The fraction of the compound that is dissolved in groundwater can be calculated from an equivalent expression:

$$f_{Dg} = \frac{\theta_g}{\theta_g + 0.41 \cdot K_{OW} \cdot f_{ocg} \cdot \rho_{bg}} \quad (A16)$$

where θ_g = volumetric water content of porous medium, t^3/t^3

f_{ocg} = organic carbon fraction of porous medium

ρ_{bg} = bulk density of porous medium, K_g/l

$$\text{and } \rho_{bg} = 2.65 \cdot (1 - \theta_g) \quad (A17)$$

The sorbed chemical fraction f_{sg} is equal to $1 - f_{Dg}$.

Slower chemical transformation reactions can be treated generally by using mixed second order kinetics (Burns et al., 1982):



where $[E]_i$ = environmental property for process "i"

P = transformation product, mg/l

The reaction rate R_i ($mg/l\text{-sec}$) for process "i" is:

$$R_i = k_i \cdot Y_i \cdot f_D \cdot C \quad (A17)$$

where k_i = second order rate constant for process "i"

Y_i = yield coefficient for process "i"

C = total concentration of compound (mg/l).

Given a local value for $[E]_i$, a pseudo-first order rate constant K_i (sec^{-1}) can be calculated:

$$K_i = k_i \cdot [E]_i \cdot Y_i \cdot f_D \quad (\text{A18})$$

For a compound undergoing several competing reactions, the overall pseudo-first order rate constant $K(\text{sec}^{-1})$ is

$$K = \sum_i K_i \quad (\text{A19})$$

This general second order reaction method can be used to predict reaction rates for photolysis, hydrolysis, oxidation, and bacterial degradation. For short reaches of rivers with travel times of hours, these reactions are not likely to significantly reduce instream concentrations. For transient loads during storms, darkness should further reduce Photolysis and, indirectly, oxidation. Bacterial communities are unlikely to acclimatize in hours to the transient loads. Of these transformation reactions, then, only hydrolysis will be considered for those few compounds with large rate constants. The nominal hydrolysis rate constant is calculated from the acid-catalyzed, neutral, and base-catalyzed pathways (Burns et al., 1982):

$$K_{HO} = (k_{HA}[H^+] (\alpha \cdot f_s + f_D) + k_{HN} + k_{HB}[OH^-] \cdot (f_D)) / 3600. \quad (\text{A22})$$

where k_{HA} = second-order acid-catalysis hydrolysis rate constant

(l/mole-hour)

$[H^+]$ = hydrogen ion concentration = $10^{-\text{pH}}$ (mole/l)

pH = stream pH

α = acid-catalysis hydrolysis rate enhancement factor for sorbed compound = 10

k_{HN} = neutral hydrolysis rate constant (sec^{-1})

k_{HB} = second-order base-catalysis hydrolysis rate constant (l/mole-sec)

$[\text{OH}^-]$ = hydronium ion concentration = $10^{-\text{pOH}}$ (mole/l)

pOH = stream pOH = $14 - \text{pH}$.

For ground water, the nominal hydrolysis rate constant (in years^{-1}) is calculated from an equivalent expression:

$$K_{go} = (k_{HA}[\text{H}^+]_g(\alpha \cdot f_{sg} + f_{Dg}) + k_{HN} + k_{HB}[\text{OH}^-]_g \cdot f_{Dg}) \cdot (24) \cdot (365.25) \quad (\text{A23})$$

where $[\text{H}^+]_g$ = hydrogen ion concentration = 10^{-pH_g} (mole/l)

pH_g = ground water pH

$[\text{OH}^-]_g$ = hydronium ion concentration = 10^{-pOH_g} (mole/l)

pOH_g = ground water pOH = $14 - \text{pH}_g$

The nominal hydrolysis rate constants K_{Ho} and K_{go} apply to a reference temperature, T_R (usually 25°C). These can be corrected to ambient surface or ground water temperatures (T or T_g) with the following expressions:

$$K_H = K_{Ho} \cdot \exp\left[10^4 \cdot \left(\frac{1}{T_R + 273} - \frac{1}{T + 273}\right)\right] \quad (\text{A24})$$

$$K_g = K_{go} \cdot \exp\left[10^4 \cdot \left(\frac{1}{T_R + 273} - \frac{1}{T_g + 273}\right)\right] \quad (\text{A25})$$

A final transformation pathway to consider is volatilization. The volatilization rate constant K_v (sec^{-1}) can be calculated from the Whitman, or two-resistance model (Whitman, 1923; Burns et al., 1982):

$$K_v = \frac{1}{d} \cdot \frac{1}{R_L + R_G} \cdot f_D \quad (A26)$$

where d = mean stream depth (m)

R_L = liquid Phase resistance (sec/m)

R_G = gas Phase resistance (sec/m).

The second term in Equation (A26) represents the conductivity of the compound through a liquid and a gas boundary layer at the water surface. The liquid phase resistance to the compound is assumed to be proportional to the transfer rate of oxygen, which is limited by the liquid phase only:

$$R_L = \frac{1}{K_{O_2} \cdot d \cdot \sqrt{32/MW}} \quad (A27)$$

where K_{O_2} = reaeration rate constant (sec^{-1})

MW = molecular weight of the compound

32 = molecular weight of oxygen.

The gas phase resistance to the compound is assumed to be proportional to the transfer rate of water vapor, which is limited by the gas phase only:

$$R_G = \frac{1}{\frac{H}{RT'} \cdot WAT \cdot \sqrt{18/MW}} \quad (A28)$$

where WAT = water vapor exchange constant (m/sec)

18 = molecular weight of water

H = Henry's law constant ($\text{atm} \cdot \text{m}^3/\text{mole}$)

R = ideal gas constant = $8.206 \times 10^{-5} \text{ m}^3 \cdot \text{atm}/\text{mol} \cdot ^\circ\text{K}$

T' = water temperature ($^\circ\text{K}$) = $273 + T$.

The reaeration and water vapor exchange constants will vary with stream reach and time of year. They can be calculated using one of several empirical formulations. The reaeration rate constant will be calculated by the Covar method using stream velocity U and depth d , then corrected for temperature (Covar, 1976). The water vapor exchange constant will be calculated using wind speed and a regression proposed by Liss (1973):

$$WAT = 5.16 \times 10^{-5} + 3.156 \times 10^{-3} \cdot W \quad (A29)$$

where W = wind speed at 10 cm above surface (m/sec).

Wind speed measured above 10 cm can be adjusted to the 10 cm height assuming a logarithmic velocity profile and a roughness height of 1 mm (Israelson and Hanson, 1962):

$$W = W_z \cdot \log(0.1/0.001) / \log(z/0.001) \quad (A30)$$

where W_z = wind speed at height z (m/s)

z = wind measurement height (m)

In summary, three transformation processes are considered in this analysis: sorption, hydrolysis, and volatilization. Sorption of hydrophobic organic compounds is calculated by Equation (A15) using data for K_{ow} , f_{ow} , and s . Hydrolysis is calculated by Equations (A22) and (A24) using data for pH , k_{HA} , k_{HN} , and k_{HB} . Volatilization is calculated by Equations (A26) through (A30) using data for U , d , W , T , MW , and H . When insufficient data are available, conservative analyses can be completed ignoring any of these processes.

APPENDIX B

ANALYTICAL SOLUTION FOR TWO-DIMENSIONAL DUE TO PULSE LOADING

The stream transport model described in the main body of this report is based on our analytical solution for two-dimensional transport from a distributed vertical plane source in uniform flow (Figure B.1). The case involving pulse release of contaminant is considered. The analytical solution for this case is developed in this appendix.

Consider the region with the Gaussian distributed source shown in Figure B.1. The advective-dispersive equation for transport of a nonconservative contaminant in uniform stream flow can be written as

$$U \frac{\partial c}{\partial x} - E_x \frac{\partial^2 c}{\partial x^2} - E_y \frac{\partial^2 c}{\partial y^2} + Kc + \frac{\partial c}{\partial t} = 0 \quad (B1)$$

where E_x and E_y are dispersion coefficients in the longitudinal, x , and transverse, y , directions, respectively, U is the main flow velocity in the x -direction, c is the solute concentration, K is the first-order decay constant, and t is the elapsed time.

The initial and boundary conditions associated with equation (B1) may be expressed as

$$c(x, y, 0) = 0 \quad (B2)$$

$$c(\infty, y, t) = 0 \quad (B3)$$

$$\frac{\partial c}{\partial y}(x, 0, t) = 0 \quad (B4)$$

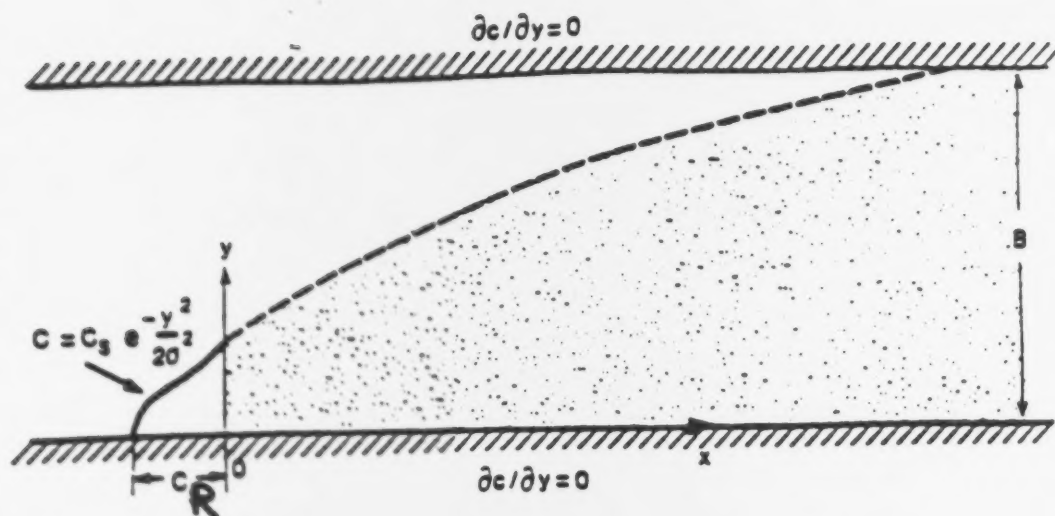


FIGURE B-1

SCHMATIC DESCRIPTION OF TWO-DIMENSIONAL TRANSPORT
IN UNIFORM FLOW IN A STREAM OF FINITE WIDTH, B .

$$\frac{\partial c}{\partial y}(x, B, t) = 0 \quad (B5)$$

$$c(0, y, t) = C_s \exp(-y^2/2\sigma^2), \quad t \leq t_R \quad (B6)$$

$$c(0, y, t) = 0, \quad t > t_R$$

where C_s and σ are the peak concentration and standard deviation of the Gaussian source assumed to be located at $x = 0$.

The analytical solution for the above case can be derived in two steps. The first step involves an application of the image source theory to the fundamental solution of the corresponding case in which the stream is of infinite width and the contaminant is continuously released from the source. Let c_f^* denote the fundamental solutions. The expression for c_f^* has been derived in the report dealing with groundwater screening procedures. It may be written as

$$c_f^*(x, y, t) = \left[\frac{\sigma_x \exp(U_x/2E_x)}{(2\pi E_x)^{1/2}} \cdot I \right] C_s \quad (B7)$$

where

$$I = \int_0^t \frac{\exp\left(-\frac{x^2}{4E_x\tau} - \frac{y^2}{4E_y\tau + 2\sigma^2} - \frac{U^2\tau}{4E_x} - K\tau\right) d\tau}{\tau^{3/2} (2\sigma^2 + 4E_y\tau)^{1/2}} \quad (B8)$$

Image sources must be applied and their effects must be added to the fundamental solution to satisfy the zero normal gradient lateral boundary conditions. (Figure B.2 the actual source plus the first four image sources.) In general, an infinite number of image sources is required to reproduce the lateral boundary effect exactly. The resulting solution becomes

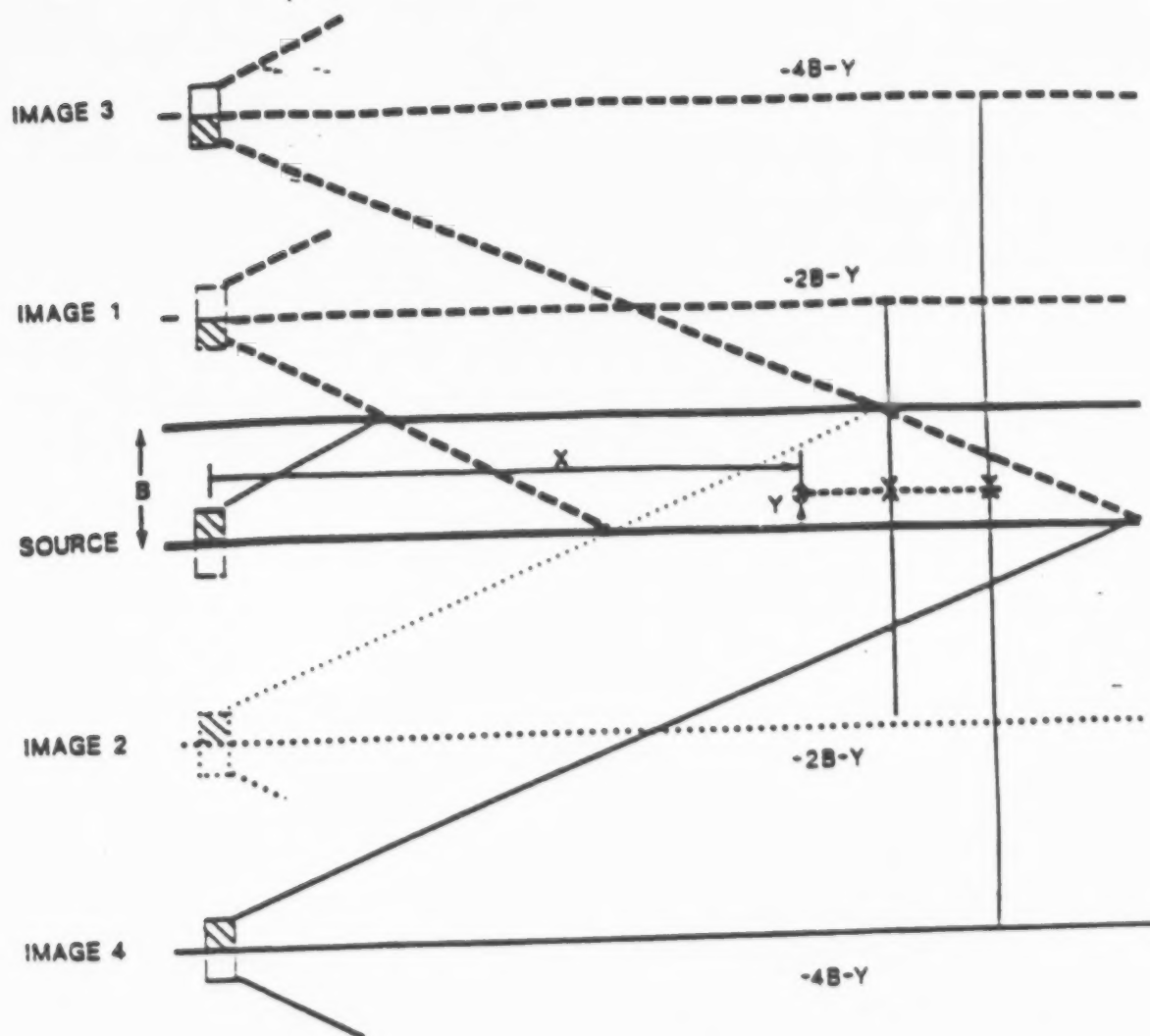


Figure 15.2
TREATMENT OF LATERAL BOUNDARY CONDITIONS
USING IMAGE SOURCES

$$C^*(c,y,t) = C_s [C_f^*(x,y,t) + \sum_{i=1}^{\infty} C_{fi}^*(x,y,t)] \quad (B9)$$

where

$$C_{fi}^* = C_f^*(x, 2Bi + y \cos(i\pi), t) \quad (B10)$$

The second step in the derivation involves an application of Duhamel's theorem of superposition in the time domain to satisfy the pulse source boundary condition (B6). Figure B.3 illustrates how the pulse load condition is obtained by superposing two continuous loads staggered over time interval t_R .

The combined response is the required analytical solution and is given by

$$\begin{aligned} C_{x,y,t} = C_s [C_f^*(x,y,t) + \sum_{i=1}^{\infty} C_{fi}^*(x,y,t) \\ - C_f^*(x,y,t-t_R) + \sum_{i=1}^{\infty} C_{fi}^*(x,y,t-t_R)] \end{aligned} \quad (B11)$$

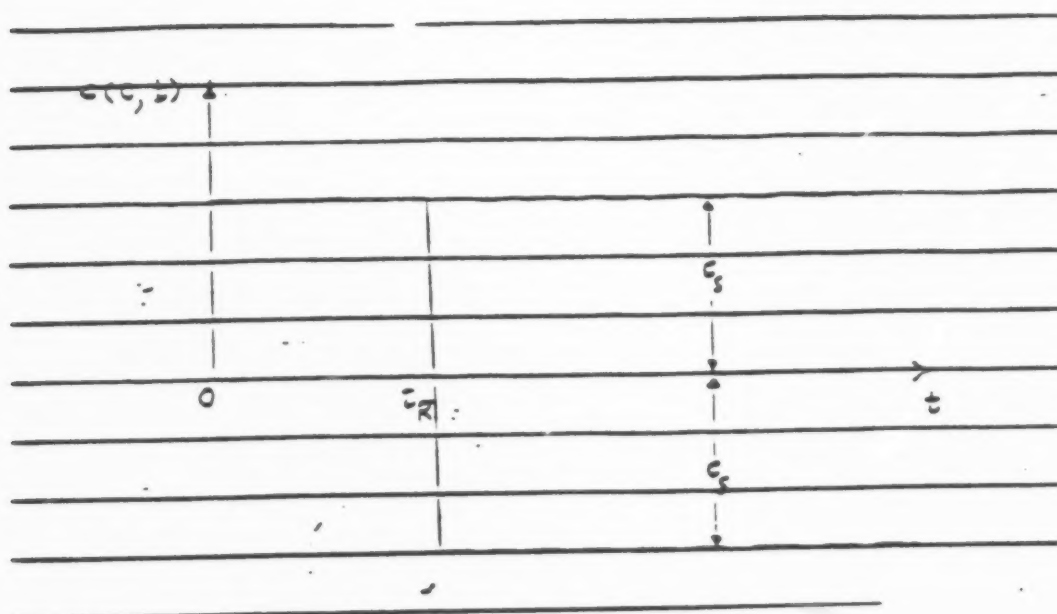


Figure 8.3. Schematic description of superposition procedure used to obtain the pulse loading condition at $x = 0$.

TECHNICAL ASSISTANCE IN THE DEVELOPMENT OF THE MONTE CARLO
UNCERTAINTY ANALYSIS FOR THE SURFACE WATER COMPONENT FOR
LAND DISPOSAL RESTRICTIONS DETERMINATIONS

DRAFT FINAL REPORT

Contract No. 68-01-7075
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1.3 Uncertainty and Its Representation in Probability Density Functions

Uncertainty in a parameter from a water quality simulation model can result from several sources. In a fundamental sense, uncertainty arises from natural variability, inadequate or unrepresentative samples, measurement error, estimation error, and probability model specification error (Reckhow and Chapra 1983). All of these are likely to exist here.

Natural variability reflects spatial and temporal differences. Certainly this source of uncertainty is characterized in the distributions presented in Section 4.0, as all of the data sets summarized in this section are cross-sectional. At issue, however, is the relationship between the natural variability in these data sets and the natural variability in the target population. For example, the available fish lipids data may represent species that fisheries biologists chose to study; this sampled population may not represent the target population (e.g., all freshwater species, weighted by their relative abundance). As a result, we may have an estimate of natural variability, but it may be inappropriate for the target population. We believe that error associated with differences between the target population and the sampled population is important in this study, but it is not reflected in the distributions in Section 4.0. This issue is discussed further in Sections 4.0 and 5.0.

Measurement error here refers primarily to laboratory precision and accuracy or to errors in the process of making an

"observation" (e.g., measuring or recording watershed area). For water quality-related parameters, it seems reasonable that multiple observations from multiple scientists/laboratories (which probably describes the data in Section 4.0) will result in the inclusion (confounding) of measurement error in the distributions. Non-water quality measurement and recording errors may exist, may or may not be confounded in the variability in the distributions, but are expected to be relatively minor in effect.

Estimation error and probability model specification error are related terms. For the Monte Carlo simulation, available information is converted into a probability model for each parameter. This task is accomplished using a combination of statistical analysis and judgment in the selection of a probability model. It is unreasonable to believe that the true model has been chosen (particularly if the sampled and target populations differ), so there is uncertainty concerning the probability model that was specified. Once the model is chosen, data are used to estimate "parameters" for the distribution. These parameters are estimates, and therefore they, too, are uncertain. Both of these sources of error exist in this study for all of the simulation model parameters, but no attempt has been made to include these errors in Section 4.0.

In conclusion, the distributions in Section 4.0 reflect natural variability and (probably) measurement error. The distributions do not reflect differences between the sampled and target populations, parameter estimation error, and probability model specification error. Of these error terms, natural

variability is relatively important, but it is also possible that one or more of the excluded errors is important. This is an area in need of further study. It should also be noted that there is another, distinct error term (the simulation model error) that should be considered herein; this issue is discussed in Section 5.0

2.0 The Simulation Model

The fish component of the steady state groundwater loading model consists of four linked stages that relate landfill leachate concentration, c_l , to the average whole fish biomass concentration, c_f (Ambrose, et al., 1985). The first stage of the model specifies that the fraction of site contaminant concentration delivered through the groundwater to the stream, C_g , is determined by the groundwater dilution factor, r_g :

$$C_g = r_g c_l \quad (1)$$

The second stage relates the average concentration in the mixing zone, c_s , to the average groundwater concentration. As in the first stage, the relationship is given in terms of a dilution factor,

$$c_s = r_s c_g \quad (2)$$

where r_s is the stream dilution factor. The third stage of the model describes the downstream routing of the contaminants from the initial mixing zone. Given the average concentration in the mixing zone, c_s , the contaminant concentration at a specified downstream point, c_x is given by

$$c_x = r_t c_s \quad (3)$$

where r_t is a dilution factor that accounts for the various processes governing the downstream transport. The final and fourth stage of the model determines the fish biomass concentration, c_f . For a fish located at a specified downstream point x , its biomass concentration is given by the product of the

dissolved concentration and whole fish partition coefficient:

$$c_f = f_d k_f c_x \quad (4)$$

where f_d is the dissolved fraction.

Combining equations (1)-(4), it can be seen that the aggregate model can be rewritten so as to specify the fish biomass concentration, c_f , as a simple fraction of the leachate concentration:

$$c_f = r_{ag} c_l \quad (5)$$

where r_{ag} , the aggregate dilution factor, is given by the product of the individual dilution factors:

$$r_{ag} = r_g r_s r_t r_f \quad (6)$$

where $r_f = k_f / f_d$. Given a value for r_{ag} and a corresponding allowable daily intake concentration standard for fish, c_{ADI} , equation (5) can be used to back-calculate the allowable waste site leachate concentration, c_l :

$$c_l = c_{ADI} / r_{ag} \quad (7)$$

Ambrose et al., (1985) derive relationships for each of the dilution factors in terms of more fundamental watershed and compound characteristics. With respect to the factors r_g and r_s , they show that their product can be approximated by:

$$r_g r_s = (A_w / A_s) f_{wh} \quad (8)$$

where A_w and A_s are the waste site and upstream watershed areas

dissolved concentration and whole fish partition coefficient:

$$c_f = f_d k_f c_x \quad (4)$$

where f_d is the dissolved fraction.

Combining equations (1)-(4), it can be seen that the aggregate model can be rewritten so as to specify the fish biomass concentration, c_f , as a simple fraction of the leachate concentration:

$$c_f = r_{ag} c_1 \quad (5)$$

where r_{ag} , the aggregate dilution factor, is given by the product of the individual dilution factors:

$$r_{ag} = r_g r_s r_t r_f \quad (6)$$

where $r_f = k_f / f_d$. Given a value for r_{ag} and a corresponding allowable daily intake concentration standard for fish, c_{ADI} , equation (5) can be used to back-calculate the allowable waste site leachate concentration, c_1 :

$$c_1 = c_{ADI} / r_{ag} \quad (7)$$

Ambrose et al., (1985) derive relationships for each of the dilution factors in terms of more fundamental watershed and compound characteristics. With respect to the factors r_g and r_s , they show that their product can be approximated by:

$$r_g r_s = (A_w / A_s) f_{ws} \quad (8)$$

where A_w and A_s are the waste site and upstream watershed areas

respectively, and f_h is the fraction of the chemical not transformed by hydrolysis or removed by speciation. An additional consideration simplifies the model further. Since the fish are assumed to reside in the mixing zone, the third stage dilution factor, r_t , can be set to unity. Finally, with respect to the fourth stage, Ambrose et al. develop a relationship for the estimation of r_f which is a function of the chemical properties of the compound and the characteristics of the target fish population:

$$r_f = k_{ow} f_d = (1 + 0.41 k_{ow} f_{oc} S 10^{-6})^{-1} k_{ow} f_l \quad (9)$$

where k_{ow} is the octanol water partition coefficient, f_{oc} is the organic carbon fraction of the sediment, S is the sediment concentration, and f_l is the fraction of fish biomass that is lipid. Substituting the appropriate terms into equation (7) gives the model version used in this study:

$$c_l = (A_s / A_w) (1 + 0.41 k_{ow} f_{oc} S 10^{-6}) / (k_{ow} f_l f_h) c_{ADI} \quad (10)$$

3.0 Sensitivity Analysis

3.1 Procedure

The objective in this section is to determine the potential for reducing the number of parameters included in the Monte Carlo simulation on the basis of the model's sensitivity to their variation. Experimental design techniques provide a systematic basis for evaluating the effects of treatments (parameters) on a given response (model output). In this study, a full factorial experimental design (Box et al., 1978) was formulated for the Drinking Water Model Steady Groundwater Loading Scenario. Six of the seven parameters specified in the model in equation (10) were considered: A_s , A_w , S , f_{oc} , f_l , and K_{ow} . The remaining parameter, f_h , since it is determined exogenously, was fixed equal to one. A total of 64 runs were made by varying each of the factors between two levels (+, -). These were determined by taking a +/-10% variation about the mean of the range of their recommended values (See Table 1). The ranges were obtained by combining the ranges recommended in the Ambrose et al. (1985) report together with the default parameter settings in the D2WATER computer model. In order to generalize the results, the factor responses were measured by the ratio of the allowable daily intake concentration and the allowable leachate concentration, c_{ADI} / c_l .

3.2 Results

The model run results are tabulated in Tables 2-3 which correspond to values of K_{ow} of $5.E4$ and $6.1E4$, respectively. Each table consists of thirty-two model runs. Each run is characterized by a series of (negative and positive) signs, which

describes the parameter combinations used in the run, and the resulting model response.

Given two runs which vary only with respect to a given factor, an estimate of that factor's effect can be obtained by subtracting the response associated with the negative factor level from that of the positive factor level. Averaging over all such contrasts provides an estimate of the main effect of varying that factor from its negative to its positive level. For example, the difference the response between runs 1 and 2, (7.59-9.24), is one estimate of the main effect of increasing A from its negative to its positive level.

These effects can be further differentiated as a function of a second treatment factor and lead to the evaluation of two factor interactions. For example, the difference between runs 17 and 18 is also an estimate of the main effect of increasing A. The latter two runs, however, were made with f at its higher level, while the former runs were made with f at its lower level. The differences in these main effect estimates, [(9.19-11.19) - (7.59-9.24)], is an estimate of the two factor interactions between A and f. In all cases, there will be thirty two such contrasts whose average then provides an estimate of the effect.

Main treatment factor effects and selected two and three factor interactions were calculated for the experimental runs reported in Tables 2-3. These results are presented in Table 4. Their investigation leads to the following conclusions. First, four of the six factors dominate the response. These include A (main treatment effect= -222%), A (222%), K (217%), and f (main treatment effect= -222%), A (222%), K (217%), and f

(217%). The main effect of the remaining two factors, S and f_{oc} , was limited to 8%. Second, the response is characterized by significant two factor interactions. As discussed in the previous section, this indicates that a factor response depends on the value of a second factor. The most important of these include $A \times A$, $A \times K$, $A \times f$, $A \times K$, and $f \times K$. Each of these results in a 21% effect. Third, compensating interactions are prominent in the results. The most evident of these involves A and A . As is evident from Tables 2 and 3, positively correlated changes that preserve the factor's ratio result in no change in response. Negatively correlated changes, however, result in significant differences in response. The same is true for the other two factor pairs presented in 2, although in general it is the negatively correlated change that produces the difference in response. This suggests that it is imperative that the correlation structure between the factors is investigated and incorporated in the monte carlo algorithm. Fifth, preliminary runs made with factors varying throughout their ranges resulted in responses that were orders of magnitude greater than the responses in Tables 2 and 3. This suggests the need for careful selection of both the distributional form and its parameters, particularly the variance. Sixth, alternative runs were made with different values of f . The response is reflected entirely by the main effect without any interactions and is strictly multiplicative. The effect of using a different value of f is obtained by multiplying the new f times the response, $C_h / C_{ADI 1}$, obtained with f equal to one. Finally, the effect of factors S and f is relatively small given the 20% change adopted here.

But caution must be exercised before deciding to eliminate them from the monte carlo simulation. Although the effects are small, the 20% variation needs to be compared with the expected variability of the parameters. In any case, however, it is not clear that the resulting computational savings in this scenario would be significant.

In summary, the analysis has pointed out the need for considering the correlation structure among the various parameters, including f , the exogenous groundwater parameter. In terms of eliminating parameters, one must be careful in the interpretation of the results since the assumed ranges in the experimental design may not reflect the variability of the parameter. In any case, the results indicate that all of the parameters are sufficiently sensitive to not warrant their elimination at this stage of the analysis.

Factor	Range		Level		Units
			(-)	(+)	
A _s	1.E7	1.E9	4.6E8	5.6E8	sq. m.
A _w	1.E4	1.E6	4.6E5	5.6E5	sq. m.
S	10.	100.	50.	60.	mg/l
f _{oc}	0.01	0.05	0.027	0.033	fract.
f _l	0.10	0.30	0.18	0.27	fract.
K _{ow}	1.E3	1.E5	5.E4	6.1E4	l /l oct wat

Table 1

	A_s	A_w	S	f_u	f_c	C_L	C_{ADJ}/C_L		A_s	A_w	S	f_u	f_c	C_L	C_{ADJ}/C_L
(+) 568569	60	033	22												
(-) 268465	50	027	18												
1	-	-	-	-	-	0.10818	9.24	17	-	-	-	-	-	0.089363	11.19
2	+	-	-	-	-	0.13169	7.59	18	+	-	-	-	-	0.10879	9.19
3	-	+	-	-	-	0.088859	11.25	19	-	+	-	-	-	0.073405	13.62
4	+	+	-	-	-	0.10818	9.24	20	+	+	-	-	-	0.089363	11.19
5	-	-	+	-	-	0.10876	9.19	21	-	-	+	-	-	0.089844	11.13
6	+	-	+	-	-	0.1324	7.55	22	+	-	+	-	-	0.10938	9.14
7	-	+	+	-	-	0.089338	11.19	23	-	+	+	-	-	0.073601	13.55
8	+	+	+	-	-	0.10876	9.19	24	+	+	+	-	-	0.089844	11.13
9	-	-	-	+	-	0.10882	9.19	25	-	-	-	+	+	0.089898	11.12
10	+	-	-	+	-	0.13248	7.55	26	+	-	-	+	+	0.10944	9.14
11	-	+	-	+	-	0.089391	11.19	27	-	+	-	+	+	0.073345	13.54
12	+	+	-	+	-	0.10882	9.19	28	+	+	-	+	+	0.089898	11.12
13	-	-	+	+	-	0.10954	9.13	29	-	-	+	+	+	0.090486	11.05
14	+	-	+	+	-	0.13335	7.50	30	+	-	+	+	+	0.11016	9.08
15	-	+	+	+	-	0.089976	11.11	31	-	+	+	+	+	0.074328	13.45
16	+	+	+	+	-	0.10954	9.13	32	+	+	+	+	+	0.090486	11.05

TABLE 2

$r_g = 1.0$ $C_{ADJ} = 1.0$

	As Aw S fac f _L	C _L	$\frac{C_{AOI}}{C_L}$		As Aw S fac f _L	C _L	$\frac{C_{AOI}}{C_L}$
(+) 5.68 5.68 60 .033 .32	- - - - -	0.089194	11.21	17	- - - - -	0.073682	13.57
(-) 4.68 4.68 50 .027 .18	+ - - - -	0.10858	9.21	18	+ - - - -	0.0897	11.15
1	- + - - -	0.073267	13.65	19	- + - - -	0.060525	16.52
2	+ + - - -	0.089194	11.21	20	+ + - - -	0.073682	13.57
3	- - + - -	0.089777	11.14	21	- - + - -	0.074164	13.48
4	+ - + - -	0.10929	9.15	22	+ - + - -	0.090286	11.08
5	- + + - -	0.073745	13.56	23	- + + - -	0.060920	16.41
6	+ + + - -	0.089777	11.14	24	+ + + - -	0.074164	13.48
7	- - - + -	0.089842	11.13	25	- - - + -	0.074217	13.47
8	+ - - + -	0.10937	9.14	26	+ - - + -	0.090351	11.07
9	- + - + -	0.073799	13.55	27	- + - + -	0.060964	16.40
10	+ + - + -	0.089842	11.13	28	+ + - + -	0.074217	13.47
11	- - + + -	0.090554	11.04	29	- - + + -	0.074805	13.37
12	+ - + + -	0.11024	9.07	30	+ - + + -	0.091067	10.98
13	- + + + -	0.074384	13.44	31	- + + + -	0.061447	16.27
14	+ + + + -	0.090554	11.04	32	+ + + + -	0.074805	13.37

$$V_g = 1.0 \quad C_{AOI} = 1.0$$

$$K_{ow} = 6.1 \times 10^4$$

TABLE 3

Factor	Response
A	-2.22
S	
As x Aw	-0.22
A x f	-0.21
S l	
A x K	-0.21
S OW	
A x K x f	-0.02
S OW l	
A	2.22
W	
A x f	0.21
W l	
A x K	0.21
W OW	
A x K x f	0.02
W OW l	
S	-0.08
S x f	-0.01
S l	
S x K	-0.01
OW	
f	-0.08
OC	
f x f	-0.01
OC l	
f x K	-0.02
OC OW	
fl	2.16
f x K	0.21
l OW	
KOW	2.17

Table 4

4.0 Probability Distributions

The sensitivity analysis in Section 3.0 provides the basis for selection of parameters examined in this Section. Based on the sensitivity analysis, probability distributions are considered for the following simulation model parameters:

Surface Area of Stream Watershed (A)
Surface Area of Waste Disposal Unit (A^S)
Fraction of Fish Biomass that is Lipid (f^W)
Octonol-Water Partition Coefficient (K^1)
Organic Carbon Content of Sediment (f^{OW})
Sediment Concentration (S^{OC})

Two changes were made following initial analysis of data. First, on a chemical-specific basis, K^{OW} is not expected to vary greatly, so the range examined in Section 3.0 may overstate the importance of K^{OW} . In the groundwater model (Mulkey and Brown 1985), K^{OW} will initially be treated as a constant and later will be varied as a function of within-laboratory precision (L. Mulkey, pers. comm.). Since the groundwater model is to provide input to the Ambrose et al. surface water model, it is proposed that this input include the chemical-specific value for K^{OW} . Thus K^{OW} will initially be represented by a chemical-specific constant and later be represented by a chemical-specific distribution; in both cases the value for K^{OW} will be determined in the groundwater model.

A second change results from the fact that f^{OC} and S appear

in the model as a product (f S) term. This product is equivalent to TOC (total organic carbon) minus DOC (dissolved organic carbon). Thus, a probability distribution is fitted for TOC-DOC in place of f S .

There are established statistical procedures for fitting probability distributions. At the same time, judgment is important in the selection of candidate distributions and in the interpretation of statistics (Benjamin and Cornell 1970). In this case, we are aware of an ongoing study (W. Ott, pers. comm.) that demonstrates that the lognormal distribution provides a good fit to environmental contaminant concentration data. Accordingly, it is our judgment that the lognormal model should be a leading candidate for all concentration parameters.

With relatively large data sets (more than 100 observations), it becomes increasingly unlikely that the nearly exact fit required by a standard goodness-of-fit test (e.g., Kolmogorov-Smirnov) for acceptance of a common parametric probability model will be found. Thus we might expect to reject all of the common models at conventional significance levels when the data set is very large. There are two options we consider when this occurs. First, noting the problems discussed in Section 1.0 on measurement error and target population versus sampled population, we question the value of finding an exact fit. If there is a model (the lognormal) that is known to fit the type of data under examination, and if the fit by eye and through examination of selected statistics appears reasonable, then is that a sufficient basis for model selection? Or, should we try

for an exact fit, using, for example, a Box-Cox transformation, or a flexible multi-parameter distribution, or a polynomial fit to the cumulative distribution function? Both approaches are employed below, so that for selected parameters, this question might be resolved with a sensitivity analysis. For example, the Monte Carlo simulation could be run, once with each candidate distribution. If there is little difference in the results, then we may be less concerned about probability model specification error.

We proceed now with a discussion and presentation of the probability distributions.

4.1 TOC - DOC (mg/l)

A STORET retrieval of TOC-DOC initially yielded several hundred thousand observations. To make the data set manageable, and to create some consistency in measurement methods and detection limits, the retrieval was finally restricted to 1983-1984. Some of the values of TOC-DOC were negative. These were due to several extremely large values (e.g., E+30) that do not appear appropriate for this analysis; hence they were eliminated from the data set.

SAS Proc Univariate was run on the untransformed and log-transformed TOC-DOC observations; the log-transformation was selected for serious consideration for reasons noted above. In the tables presented below, we focus on the (Kolmogorov) D statistic, the skewness, the difference between the mean and median, and the relative size of the quartiles as measures of

distribution fit and symmetry. For the first table, the D statistic indicates a normal distribution is unlikely. This is confirmed by the high skewness, the relatively large difference between the mean and median, and the fact that the third quartile is much larger than the second quartile (another measure of asymmetry).

The second table concerns log-transformed TOC-DOC, and thus provides an evaluation of the lognormal model. The D statistic is low which is encouraging for lognormal fit; however, it is not low enough (due to the large sample size) to lead to acceptance of lognormality at the 0.01 level. Skewness is relatively small, the mean is quite close to the median, and the third quartile is similar in size to the second quartile. All of these suggest symmetry. Thus, given the prior evidence of lognormality, the relatively low value of Kolmogorov's D, and the evidence concerning symmetry, we recommend for TOC-DOC:

Lognormal pdf

Mean = -0.19

Variance = 1.90

TABLE BA

15:39 SATURDAY, OCTOBER 26, 1985

DATA REQUEST FOR NEW PECKHAM/VEDIN PINES
PREPARED BY DFWA
DATA SOURCE: STURF

UNIVARIATE

VARIABLE=TIME DIFFERENCE: TIME-DISSEMINATED

MEASURES

N	743	SUP: 635	743
MEAN	2.038	SUP: 1.00	2.12706
STD DEV	21.4578	VARIANCE	4.6100
SKEWNESS	1.02166	KURTOSIS	201.223
KURTOSIS	36.060	COV	354.006
CV	745.273	STD PLAN	0.001706
TEMPERATURE	3.00237	PROBABILITY	0.002060
TEMPERATURE	11.655	PROBABILITY	0.00001
TEMPERATURE	505	PROBABILITY	0.00001
TEMPERATURE	0.440569	PROBABILITY	0.00001

CUMULATIVE

100% PAX	500	992
75% PAX	1.3	952
50% PAX	0.5	902
25% PAX	0.1	102
12.5% PAX	0	52
6.25% PAX	0	12
3.125% PAX	0	12
1.5625% PAX	0	12
0.78125% PAX	0	12
0.390625% PAX	0	12
0.1953125% PAX	0	12
0.09765625% PAX	0	12
0.048828125% PAX	0	12
0.0244140625% PAX	0	12
0.01220703125% PAX	0	12
0.006103515625% PAX	0	12
0.0030517578125% PAX	0	12
0.00152587890625% PAX	0	12
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TABLE 5b

DATA REQUEST FOR NEW PECKUM/EYDIE PINES
PREPARED BY DHR
DATA SOURCE: ST0611

15:36 SATURDAY, OCTOBER 26, 1985

UNIVARIATE

VARIABLE=LN_DIFF LEFT=REFERENCE

REPORTS

N 505
MEAN -0.14311
STD DEV 1.3772
SKEWNESS -0.147217
KURTOSIS 1146.91
CV -711.619
T-TEST=0
SIGN DIFF -14776.2
NORM Z=0
D: PEARSON 0.0870266

QUANTILES(DEF=4)

100% MAX 6.21461
75% Q3 0.470001
50% MED -0.23143
25% Q1 -0.516292
0% MIN -13.8629
RANGE 20.0776
Q3-Q1 1.38629
FIDF -1.20397

EXTREMES

LOWEST
-13.8629
-2.30259
-2.30259
-2.30259
-2.30259
-2.30259

4.11278
2.20378
1.24683
-1.60944
-2.30256
-2.30259

992
952
902
102
52
12

HIGHEST
4.11278
4.40511
5.24702
5.35361
6.21461

MISSING VALUE
COUNT 3555
% COUNT/TOTALS 85.60

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4.2 Distribution for A (square miles)

The data set for A consisted of 39,679 observations of adjusted area, total area, and stream order number for upstream watersheds throughout the continental U.S. As expected, the frequency of the data descends with order number: 51% (30,176) first order, 24% second order, 13% third order, 7% fourth order, 4% fifth order, 1% sixth order, and <1% seventh order. Two major difficulties were faced in this analysis: 1) the lack of paired waste site/watershed area observations, and 2) extremely large sample sizes.

Given the unavailability of paired watershed/waste site area observations, the choice of sampling scale (total area, adjusted area, first order stream) becomes problematic. Ideally, the choice of scale is determined by the objectives of the experiment. In the absence of a complete data set, however, consideration is instead limited to the inferences that are possible with the available data.

For this case, two issues are relevant to choice of scale: 1) the correlation between the number of waste sites and watershed (order); and 2) the limits of applicability of the predictive model. In the first instance, the lack of data forces the assumption of a single waste site (unit of measurement in the waste site data base) per watershed area. But one would expect an increase in the number of waste sites as the watershed area increases. A random sample of total areas, which would be characterized by a very large range of watershed areas, would be

incompatible with the single site assumption. Alternatively, a random sample of adjusted areas would be incompatible with the model assumptions since the diluting volume contribution of the upland lower order streams would not be taken into account. A first order stream sampling scheme on the other hand, while it can not surmount the inadequacies of incomplete data sets, does provide the least problematic alternative: 1) it is compatible with the dilution model formulation; 2) it focuses on the more serious impacts; and 3) it is more compatible with the single waste site assumption than the total area sampling scheme.

The very large sample size resulted in additional difficulties. First, it made it cumbersome and expensive to perform computationally intensive analysis. Second, it made the application of traditional goodness of fit hypothesis tests problematic since the slightest variation from the observed cdf (cumulative distribution function) resulted in significant values for the Kolmogorov D statistic especially given the highly skewed (skewness coefficient = 18.3!) and variable (coefficient of variation = 877.5) nature.

Two criteria were used in guiding model selection: (1) Kolmogorov-Smirnov D statistic (maximum difference between the observed and predicted cdf), and (2) RMSE, the root mean square difference between the observed and predicted cdf's. Given the broader methodological concerns about the overall framework, a 2-parameter lognormal model was fitted to the first order watershed area, both for consistency and ease of analysis. The relevant distribution is,

$$\ln(A_s) \sim N(3.64, 0.94)$$

(11)

with $D=0.046$, $RMSE=0.023$ (see figure 1). Further refinement could be undertaken and could include fitting Tukey lambda or Pearson type distributions. However, preliminary runs should first be undertaken to provide an indication of the importance of model specification.

4.2.1 Random placement of waste site on watershed area A_s .

Because the waste site on a first order stream will not always occur at the mouth of the watershed, a scheme was developed for randomly allocating a waste site along the streams length. If a first order stream/watershed can be modeled by an isosceles triangle (see figure 2) with sides given by b and base equal to a , with perpendicular bisector of length L , then its area A_s is simply aL . If a waste site is randomly located $u\%$ of the length from the mouth along the bisector L , so that the waste site's distance to the mouth is uL , then the upland watershed area, A'_s , is $A_s (1-u)^2$. Thus, if u is a uniform random number on the interval $(0,1)$, the random placement of a waste site along a first order stream channel can be modeled by using the upland watershed area, A'_s ,

$$A'_s = A_s (1-u)^2 \quad (12)$$

4.3 Distribution for A_w (square miles)

The waste site data consists of 280 observations of landfill, waste pile, surface impoundment, and land treatment areas from the RIA and Part B data bases. A number of models were

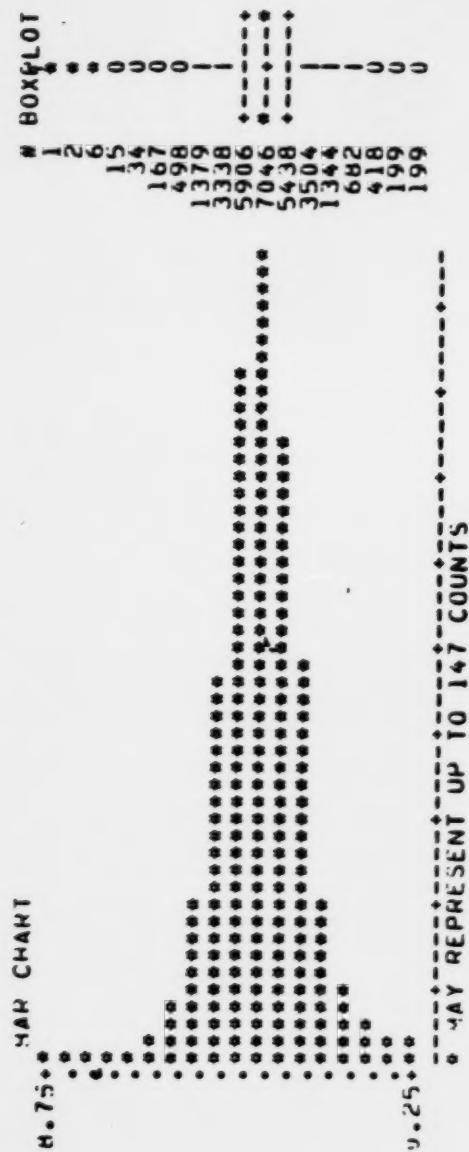


Figure 1. Frequency Plot of LN(A₃)

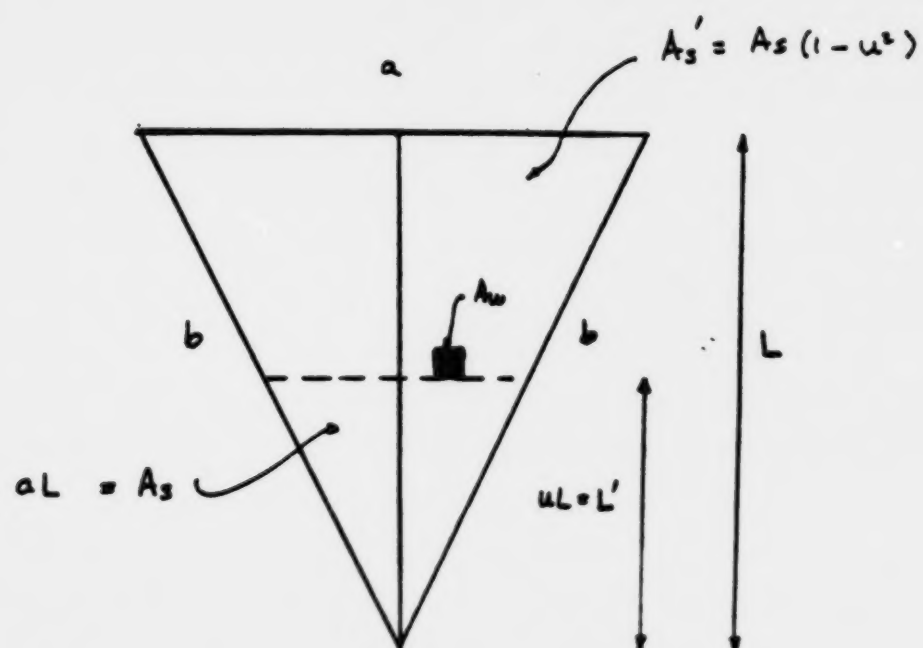


FIGURE 2

investigated including the Gamma and 2 and 3 parameter lognormal distributions. The data analysis suggests that the data follow a lognormal distribution. In this case the hypothesis that the distribution is not 2-lognormal cannot be rejected at the 15% probability level. The relevant distribution is

$$\ln(A) \sim N(-6.49, 6.32) \quad (13)$$

w

with $D=.037$, $RMSE=0.014$ (see figure 3).

4.4 Fish Lipid Fraction (f_1)

This parameter provides a good opportunity to discuss possible differences between the target population and the sampled population. If there are differences in f_1 between species, and if the relative abundance of species differs among sites, then available data on f_1 alone will be inadequate for parameter estimation.

Ideally, we might conduct a watershed-specific study (see Section 5.0). In that case, we would have information, for each watershed in the sample, on the species of fish present in the receiving water body and the relative abundance of each species. This information would be combined with the species-specific f_1 to yield a probability distribution for f_1 . Of course, this analysis was not undertaken here, so the analysis presented below represents unweighted, available f_1 data.

Percent lipid content data was obtained from STORET files. The data set consisted of 705 data points for fresh water species. Preliminary data analysis suggests that the 2-parameter



FIGURE 3. FREQUENCY PLOT OF $\ln(A_w)$

model provides a reasonable representation of the data. The relevant distribution is (see figure 4):

$$\ln(f_1) \sim N(-3.12, 0.903) \quad (14)$$

with $D=0.03$ and $RMSE=0.017$. Again as before, the importance of the choice of distribution must be evaluated before more refined procedures are warranted.

4.5 Post Sensitivity Analysis

In this section, an analytical solution of the probability and cumulative distributions of the ratio c_1 / c_{ADI} is derived and is used to test the potential effect of ignoring the variability in the term $Sx f_{oc}$.

Assuming that the term $(1 + .41 K_{ow} S f_{oc} 10^{-6})$ can be set equal to one, and ignoring for the moment the random placement of waste sites along the watershed area, then for r_h equal to unity, the $\ln$ of the ratio of leachate standard to the ADI standard, z , can be represented as

$$z = \ln(c_1 / c_{ADI}) = \ln(A_s) - \ln(A_w) - \ln(k_{ow}) - \ln(f_1) \quad (15)$$

Since each of the random terms is lognormally distributed and the term k_{ow} is, for a given chemical, a constant (say $5.e4$), then it follows that z is normally distributed with mean 2.34 and variance 8.14;

$$z \sim N(2.34, 8.14) \quad (16)$$

The cumulative distribution function of z , evaluated at any point z_0 , $F(z_0)$, can be interpreted as the percent of waste sites that will violate the c standard if all sites are in compliance with a leachate standard c_1 . For any given F value, the corresponding z value can be calculated as:

$$z = \exp(2.34 + k \cdot 2.85) \quad (17)$$

where 2.85 is the square root of 8.14 and k is given by:

$$k = [F^{.137} - (1-F)^{.137}] / .1998 \quad (18)$$

Representative values of the cumulative distribution are presented in Table 6 where it can be seen, for example, that setting the leachate standard to 27% of the ADI standard will result in violation of the ADI standard in 10% of the sites.

To test the importance of ignoring the S_{xf} term, 1000 monte carlo simulations of equation (10) were performed. The resulting distribution of z was normal with mean 2.49 and variance 8.17. The closeness of the agreement with the analytical solution, as further evidenced by the monte carlo derived cumulative in Table 7, suggests not only that the S_{xf} does not warrant very detailed examination but also that the analytical solution can serve as an acceptable approximation in lieu of a complete monte carlo solution.

$F(z)$	0.01	0.05	0.10	0.25	0.50	0.75	0.90	0.95	0.99
z	.013	.094	.267	1.52	10.4	70.7	403.	1150	8122

Table 6

F(z)	0.01	0.05	0.09	0.23	0.49	0.72	0.89	0.95	0.99
z	.013	.094	.267	1.52	10.4	70.7	403.	1150.	6122

Table 7

5.0 Concluding Observations

The Environmental Protection Agency faces a difficult task in the regulation of the land disposal of hazardous substances on a generic basis. Clearly, there are chemical-specific and site-specific factors that are important. Since the Monte Carlo simulation will be run on a chemical-by-chemical basis, we focus here on a discussion of site-specific considerations. Beyond that, however, we raise some questions concerning the approach proposed. This is done in the belief that continual questioning and re-examination of the approach taken by EPA will ultimately result in the most defensible regulations.

1. All simulation models describing natural processes contain some uncertainty in the equations (specification error). This is simply due to the fact that scientific knowledge is limited, and not all scientific knowledge is incorporated into a mathematical model. For a particular process or system, we might expect that simple models (e.g., those with few parameters) contain relatively more error in the equations and less in the parameters. In part this is due to the fact that simple models aggregate and simplify behavior.

The Ambrose et al. model is relatively simple, and thus it can be expected to have a non-negligible model error. At this point, we have no basis for estimating this error, but it is certainly possible that its effect on the model response variable would be comparable to the effect of one of the important

parameters. We recommend that an attempt be made to estimate model error, either subjectively, or using site-specific data (see below), or using information from both sources in a Bayesian framework.

2. For all parameters, we must be concerned about the equivalence (or lack of equivalence) between the sampled population and the target population. In some cases (see Section 4.0), it may be possible to describe the difference between these populations, but it is difficult to quantify the effect of this difference on parameter uncertainty. If this difference is important (we think that it could be), then an alternative approach might be considered.

We propose, therefore, that a sample be drawn to insure that the sampled and target populations are equivalent. For example, since the analysis is fundamentally concerned with a disposal site located (now or in the future) in an upland watershed, a sampling frame for upland watersheds in the U.S. could be identified. From this sampling frame, a probability sample (either random or stratified random, with strata to be defined) of watersheds could be drawn that would allow inferences to the population of watersheds (that do contain, or might contain, a land disposal site).

For each watershed in the sample, a complete set of watershed-specific parameter estimates would be determined. These might be found in existing data bases using geographic proximity as a selection criterion. Alternatively, some new data collection

may be required. The result would be, for example, a distribution for the fish lipids parameter that would reflect the relative abundance of species found at the site.

What are the advantages to this approach? At least three can be identified: (1) the target population and the sampled population are equivalent (thus the data will represent what it is intended that they represent), (2) the site-specific data provide the opportunity to estimate parameter covariances (see below), and (3) the site-specific data can also provide an opportunity to estimate model error. Concerning this latter point, it is unlikely that model error can be satisfactorily estimated, given the data necessary to evaluate the model for a variety of chemicals, sites, and conditions. However, an observation-prediction comparison could be made with site-specific data. Even if this occurred on a limited scale, it would provide the basis for determining the importance of model error.

3. The distributions in Section 4.0 have no association with each other that would permit the estimation of parameter correlations. In fact, there appears to be little objective basis for determining most of these correlations. Judgment could always be used, but this can be difficult and unreliable. For example, what is the correlation between the lipids parameter and the watershed area? While it may appear that there is no association, one must consider that stream order may covary with both and thus cause the two to be correlated.

Thus, it is assumed herein that the parameters are independent, even though this is almost certainly not true.

Parameter correlation can either increase or decrease the error in the model response variable, sometimes significantly so. Site-specific data sets would permit the estimation of parameter correlations, simply through an analysis of the data. It is not unreasonable to think that the effect of parameter correlations will be comparable to that of an important parameter; therefore some consideration should be given to their estimation. As a final point, we note that correlation between parameters in the groundwater model (Mulkey and Brown 1985) and the surface water model (Ambrose et al. 1985) is also possible and should be considered.

REFERENCES

- Ambrose, R.B., L. Mulkey, and P.S. Huyakorn, "Methodology for Assessing Surface Water Contamination Due to Land Disposal", EPA draft report, 1985.
- Benjamin, J.R. and C.A. Cornell, Probability Statistics and Decisions for Civil Engineers. McGraw-Hill, 1970.
- Box, G.E.P., W.G. Hunter, and J.S. Hunter, Statistics for Experimenters, J. Wiley, 1978.
- Box, G.E.P., and D.R. Cox, "An Analysis of Transformations", J. Royal Statistical Society, Series B, v26, no. 211, 1964.
- Mulkey, L.A., and L. Brown, Development of Land Disposal Banning Decisions Under Uncertainty, EPA Report, 1985.
- Reckhow, K. H., and Chapra, S. Engineering Approaches for Lake Management, Vol. 1, Ann Arbor Sc., 1983.

Proposed Surface Water Screening Levels

PROPOSED SURFACE WATER SCREENING LEVELS

<u>CHEMICAL</u>	<u>WASTE CODE</u>	<u>WATER QUALITY CRITERIA, mg/l</u>	<u>HYDROLYSIS CONSTANT/YR</u>	<u>LOG K_{ow}</u>	<u>SCREENING LEVEL, mg/l</u>
penta- chloro- phenol	F027	0.003	0	5.06	1.0
1,1,2- trichloro- 1,2,2- trifluoro- ethane	F002	0.833	>10Y*	3.10	320

* Base Hydrolysis



TETRA TECH, INC.
A HONEYWELL SUBSIDIARY
3748 MT DIABLO BOULEVARD
LAFAYETTE, CALIFORNIA 94549
SUITE 300 (415) 283-3771

December 30, 1985
File No. TC-3079

Ms. Eydie Pines
U.S. Environmental Protection Agency
(WH 562 B)
401 M Street, Southwest
Washington, D.C. 20460

Re: Parameter distributions and equations used in the Monte Carlo analysis of
the surface water models.

Dear Ms. Pines:

On December 24, 1985, I sent a computer tape containing my Monte Carlo
simulation routine for analyzing the surface water models. I sent the tape to
Dr. Lee Mulkey at the Athens, GA, EPA lab. This routine is called EPA/SURFACE/MCARLO.

In order to clarify the mathematical basis of this routine, I've enclosed
three tables listing the equations and parameter distributions used. Table 1
shows the two surface water models (Scenario's 1b and 1c) and the 1-D groundwater
model. Note that the only difference between scenario 1b and 1c is the whole
fish partition coefficient. The 1-D ground water model only takes into account
hydrolysis. Table 2 lists the parameters and their distributions used in the
surface water models. The mean and standard deviation values given in Table 2
represent the final values given to me on my December 24, 1985 telephone conversation
with Dr. Carlos Marin from Duke University. Table 3 lists the parameters and
their distributions used in the ground water model.

Please feel free to call me for any additional assistance at (415) 283-3771.

Sincerely,

Michael J. Ungs
Senior Engineer
Environmental Systems Engineering

MJU/np
Enclosure

cc: Bob Ambrose
Carlos Marin
Lee Mulkey
Ken Reckhow
Gregg Snell

Table 1
MODELS USED IN MONTE CARLO ANALYSIS

Surface Water Models

1. Scenario 1b

$$c_{std}/c_{adi} = A_s (1 + .41 K_{ow} * f_{oc} S * 1 \times 10^{-6}) / (A_w * K_f * R_g)$$

where

$$K_f = K_{ow} * f_1 \text{ (whole fish partition coefficient)}$$

2. Scenario 1c

$$c_{std}/c_{adi} = A_s (1 + .41 K_{ow} * f_{oc} S * 1 \times 10^{-6}) / (A_w * R_g)$$

Groundwater Model

$$R_g = \exp (-K_h * R_d * x/V_x) \text{ (groundwater dilution term)}$$

where

$$R_d = 1 + \rho_b * K_d / \theta \text{ (retardation coefficient)}$$

K_h = hydrolysis rate constant, adjusted for temperature and pH

$$K_h = \frac{(K_{hao} * 10^{-pH} * (\alpha * f_s + f_d) + K_{hno} + K_{hbo} * 10^{-pOH} * f_d) * A_{rhnu}}{A_{rhnu}}$$

$$f_d = 1/R_d$$

$$f_s = 1 - f_d$$

$$A_{rhnu} = \exp (1 \times 10^4 * (1/(T_{ref} + 273) - 1/(T + 273)))$$

Table 2

SURFACE WATER PARAMETERS*

<u>Parameter</u>	<u>Distribution Type</u>	<u>Distribution Parameters</u>		<u>Range min-max</u>
		<u>mean</u>	<u>std.dev.</u>	
A_s (mi ²)	Log _e normal	3.64	.97	---
U^{**}	Uniform	---	---	(0-1)
A_w (mi ²)	Log _e normal	-6.49	2.51	---
$f_{oc}S$ (mg/l)	Log _e normal	-0.19	1.38	---
f_1 (%)	Log _e normal	-3.12	.95	---

**Watershed area is modified by a "triangular" distribution₂ to account for the distance of a disposal site along a stream: $A_s (1 - U^2)$

*Reckhow, K., C. Marin and R. Giulio. 1985. Technical Assistance in the Development of the Monte Carlo Uncertainty Analysis for the Surface Water Component for Land Disposal Restrictions Determinations. Duke University, School of Forestry and Environmental Studies.

Table 3
GROUND WATER PARAMETERS*

Parameter	Distribution Type	Distribution Parameters		Range min-max
		mean	std. dev.	
Temperature (°C)	Normal	14.4	5.29	0-30
pH	Normal	6.2	1.28	.3-14
f _{oc} (%)	Log _e normal	-5.76	3.17	.001-.01
d (m)	Log ₁₀ uniform	--	--	4x10 ⁻⁶ -1x10 ⁻³
S (m/m)	Exponential	.0309	--	-- --
V _x (m/yr)	Derived from d, S	--	--	.01-9250
B (m)	Exponential	78.6	--	3-560
H (m)	Uniform	6.	--	2-10

*Mulkey, Lee and L. Brown. 1985. Development of Land Disposal Banning Decisions Under Uncertainty. Environmental Research Laboratory, U.S. EPA, Athens, Georgia.

Table 1

MONTE CARLO SIMULATION RESULTS OF THE SURFACE MODEL* FOR SCENARIO 1c

Compound	$\log_{10}(K_{ow})$	$K_{hbo}(25^{\circ}C)$ [1/(molar*hr)]	C_{std}/C_{adi} Protection Level		
			85%	90%	95%
<u>Pentachlorophenol</u>	5.06	0	770	<u>366</u>	117
Chlorobenzene	2.87	1×10^{-5}	759	388	140
Cresols	2.15	0	700	342	109
Toluene	2.82	0	701	342	109
1,2-Dichlorobenzene	3.58	1×10^{-4}	764	389	140
<u>Freon</u>	3.10	7.9×10^{-6}	761	<u>388</u>	140
Trichlorofluoromethane	2.52	1.1×10^{-8}	759	387	140

*6000 simulation runs each, using a fixed "seed"

x = 150 meters longitudinal distance

1-D groundwater model with no dispersion; only the dissolved portion is considered

 $K_{hao} = K_{hno} = 0$ hydrolysis coefficients (at a reference temperature of $25^{\circ}C$)Watershed area = $A_s(1-U^2)$ where U is uniformly distributed(0,1)



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A HANFORD, CALIF. SUBSIDIARY
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December 30, 1985
File No. TC-3079

Ms. Eydle Pines
U.S. Environmental Protection Agency
(WH 562B)
401 M Street, Southwest
Washington, D.C., 20460

Re: Final Monte Carlo Results of Seven Chemical Compounds

Dear Ms. Pines:

Enclosed you will find the final Monte Carlo simulation results for the seven chemical compounds you gave to me over the phone on December 23, 1985. Table 1 lists the compounds, its octanol-water partition coefficient, the base catalyzed hydrolysis coefficient and the resulting protection levels.

Note that the conversion between half life T_h and hydrolysis coefficient K_h is as follows:

$$0.5 = \exp(-K_h \cdot T_h) \quad 1)$$

Hence,

$$K_h = -[\log_e(0.5)]/T_h \quad 2)$$

Please feel free to call me for any additional assistance at (415) 283-3771.

Sincerely,

Michael J. Ungs
Senior Engineer

MJU:csn
Enclosure

COMPARISON OF AQUATIC TOXICITY EFFECT LEVEL
TO THE REFERENCE DOSE FOR SOLVENTS AND DIOXINS

Prepared by

Environmental Research Laboratory
U. S. Environmental Protection Agency
Duluth, Minn.

Prepared for

Office of Solid Wastes
U. S. Environmental Protection Agency
Washington, D.C.

September, 1985

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY

DATE Environmental Research Laboratory-Duluth, 6201 Gongdon Boulevard
September 13, 1985 Duluth, Minnesota 55804

SUBJECT Aquatic Toxicity of Fast Track and California Wastes

FROM Nelson A. Thomas *N. A. Thomas*
Chief, Water Quality Research Branch
ERL-Duluth, Duluth, MN

THRU: Norbert A. Jaworski, Ph.D.
Director, ERL-Duluth
Duluth, MN

TO Nancy Zeller
Characterization and Assessment Division (WH-562B)

A comparison of aquatic toxicity effect level to ADI indicates that most of the wastes on the Fast Track and California lists will be driven by the environmental number. On the attached tables I have indicated the aquatic no effect value when available from a criteria document. I have only included those where the aquatic effects occur at lower concentrations than the ADI. I have added a separate table when the aquatic effects is higher than the ADI. The 85 criteria are available now. The 86 criteria will be available in draft form in November 1985 and the final shortly after January 1, 1986. The 87 criteria will be available October 1, 1986 and the final January 1987.

Where the source of data for the comparison was AQUIRE, I used the lowest LC50. To protect against mortality the number should be divided by 2. The no effect concentration would be 5 to 100 times less.

I am still evaluating the U136 Cacodylic Acid, P03R Diethylarsine, P056 Fluorine, U150 Melphalan, U160 Methyl Ethyl Ketone Peroxide, and P096 Phosphine. It may be necessary to conduct a 96-hr acute test on some of these where we do not have data or cannot predict the toxicity from QSAR. The compounds that have an "E" behind the value are from QSAR.

Suggested possible criteria documents include Toluene and Cresols. Acute tests should be conducted on X001 1,2,2 Trichloro-1,2,2 Trifluoroethane, U212 2,3,4,6-Tetrachlorophenol and U121 Trichloromonofluoromethane.

I will continue to refine my comparison and keep you informed.

In reply to your inquiry for a comparison of Log P values. The values provided by OSW were generated by Dr. Gilman Veith so they would still be the same.

We will provide you with biodegradation half-life numbers versus BOD numbers and bioconcentration factors in the very near future.

			AQ	ADI	
U037	Chlorobenzene	108-90-7	.02	1.0	87 criteria 9/86
U052	Cresols	1319-77-3	.1	1.8	AQUIRE
U070	1,2-Dichlorobenzene	95-50-1	.36	3.15	1,2,4-Trichlorobenzene 87
U242	Pentachlorophenol	87-86-5	.003	1.0	86 criteria Final
U220	Toluene	108-88-3	.61	15.	AQUIRE
U230	2,4,5-Trichlorophenol	95-95-4	.45	3.5	87 9/86
U008	Acrylic Acid	79-10-7	.15	2.9	AQUIRE
X002	Cadmium	7440-43-9	.00066	.005	85 criteria
X003	Chromium	1333-82-0	.011	.12	85 criteria
P063	Hydrogen Cyanide	74-90-8	.005	.8	85 criteria
X005	Lead	7439-92-1	.0013	.015	85 criteria
U144	Lead Acetate	301-04-2	.0013	.015	85 criteria
P092	Phenyl Mercuric Acetate	62-38-4	.000012	.003	85 criteria
P106	Sodium Cyanide	143-33-9	.005	1.4	85 criteria
P030	Soluble Cyanide Salts	57-12-5	.005	.7	85 criteria

Based on Most Toxic Component - Underlined

P013	<u>Barium Cyanide</u>	542-62-1	.005	2.45	85 criteria
U032	<u>Calcium Chromate</u>	13765-19-0	.00029	.12	85 criteria
P021	<u>Calcium Cyanide</u>	502-01-8	.005	1.3	85 criteria
P033	<u>Chlorine Cyanide</u>	506-77-4	.005	1.8	85 criteria
P029	<u>Copper Cyanide</u>	544-92-3	.005 HCN	2.6	85 criteria
U145	<u>Lead</u>	7446-27-7	.0013	.015	85 criteria
U146	<u>Lead</u>	1335-32-6	.0013	.015	85 criteria
P065	<u>Mercury Fulminate</u>	628-86-4	.000012	.1	85 criteria
X004	<u>Nickel</u>	13981-37-8	.056	.35	slightly soluble
P073	<u>Nickel</u>	13463-39-3	.056	On test	80 criteria 86 9/30/85
P074	<u>Nickel Cyanide</u>	557-19-7	.005 HCN	On test	86 criteria 86 9/30/85
P099	<u>Potassium Silver Cyanide</u>	506-61-6	.005 HCN	5.6	85 criteria
U205	<u>Selenium Disulfide</u>	7488-56-4	.5	.	87 criteria 9/30/86
P104	<u>Silver Cyanide</u>	506-64-9	.005 HCN	3.9	85 criteria
X001	1,2,2 Trichloro-	76-13-1	.833E	956	86 criteria
U212	1,2,2 Trifluoroethane				QSAR
U121	2,3,4,6-Tetrachlorophenol	58-90-2	.01 .003E	.	QSAR
X010	Trichloromonofluoromethane	75-69-4	6.48E	12.	QSAR
P010	Arsenic	7440-38-2	.019	.05	85 criteria
P012	Arsenic Acid	7778-39-4	.019	.05	85 criteria
P011	Arsenic III Oxide	1303-28-2	.019	.05	85 criteria
U204	Arsenic U Oxide	1327-53-3	.019 As III	.05	85 criteria
P103	Selenious Acid	7783-00-8	.5	5.6	96 criteria
P121	Selenourea	630-10-4	.5	.	
	<u>Zinc Cyanide</u>	557-21-1	.005 2		86 criteria
			HCN		85 criteria
0188	Phenol	108-45-2	.01	3.5	AQUIRE
P121	<u>Zinc Cyanide</u>	557-21-1	47 Zn		80 criteria 86 9/30/85
			.005 HCN		85 criteria

WASTES WHERE THE ADI IS HIGHER
THAN AQUATIC EFFECTS

<u>Wastes</u>	<u>Aquatic Effect</u>	<u>ADI</u>
P022	100	.18
U211	.9	.0003
U140	22	.
U159	120	1.6
U226	11	1.
U080	17	.0029
U169	.46	.16
U196	3.5	.08
U210	1.0	.07
U228	.1	.003
U231	.1	.002
P006	.025*	.01
P031	731E	1.5
U123	151	70
U135	.004	.05
U151	.01	.003
P076	- Nitrogen	3.5
P078	- Nitrogen	35.
P098	.001	1.9
U201	.5	.
P103	.5	.
P108	.87	.008
P110	.02	.000004
P113	.33	.015
U215	.33	.016
U217	3.0	.018
U214	3.0	.018
U214	3.0	.019
P115	8	.017

*Still being evaluated.

WATER QUALITY CRITERIA

federal register

**Monday
July 29, 1985**

Part II

Environmental Protection Agency

**Water Quality Criteria; Availability of
Documents; Notice**

federal register

**Tuesday
November 8, 1983**

Part II

**Environmental
Protection Agency**

Water Quality Standards Regulation

ENVIRONMENTAL PROTECTION AGENCY

(OW-FRL-2871-8)

Water Quality Criteria; Availability of Documents

AGENCY: Environmental Protection Agency.

ACTION: Notice of final ambient water quality criteria documents.

SUMMARY: EPA announces the availability and provides summaries of nine ambient water quality criteria documents and national guidelines for criteria development. These criteria are published pursuant to section 304(a)(1) of the Clean Water Act. These water quality criteria may form the basis for enforceable standards.

Availability of documents: This notice contains: (1) Summaries of nine documents containing final ambient water quality criteria for the protection of aquatic organisms and their uses, (2) a summary of changes in the document entitled "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" (which is an updated and revised version of the Guidelines previously published at 45 FR 79341; November 28, 1980), and (3) responses to public comments on the Guidelines. Copies of the complete criteria documents and the revised Guidelines may be obtained from the National Technical Information Service (NTIS), 5285 Port Royal Road, Springfield, VA 22161 (phone number (703) 487-4850). A list of the NTIS publication order numbers for all 10 documents is published below. These documents are also available for public inspection and copying during normal business hours at: Public Information Reference Unit, U.S. Environmental Protection Agency, Room 2404 (rear), 401 M Street SW., Washington, D.C. 20460. As provided in 40 CFR Part 2, a reasonable fee may be charged for copying services. Copies of these documents are also available for review in the EPA Regional Office libraries. Copies of the documents are not available from the EPA office listed below. Requests sent to that office will be forwarded to NTIS or returned to the sender.

1. Ambient Water Quality Criteria for Ammonia—EPA 440/5-84-001; NTIS Number PB85-227114

2. Ambient Water Quality Criteria for Arsenic—EPA 440/5-84-033; NTIS Number PB85-227445

3. Ambient Water Quality Criteria for Cadmium—EPA 440/5-84-032; NTIS Number PB85-227031

4. Ambient Water Quality Criteria for Chlorine—EPA 440/5-84-030; NTIS Number PB85-227429

5. Ambient Water Quality Criteria for Chromium—EPA 440/5-84-029; NTIS Number PB85-227478

6. Ambient Water Quality Criteria for Copper—EPA 440/5-84-031; NTIS Number PB85-227023

7. Ambient Water Quality Criteria for Cyanide—EPA 440/5-84-028; NTIS Number PB85-227460

8. Ambient Water Quality Criteria for Lead—EPA 440/5-84-027; NTIS Number PB85-227437

9. Ambient Water Quality Criteria for Mercury—EPA 440/5-84-028; NTIS Number PB85-227452

10. Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses. NTIS Number PB85-227049.

FOR FURTHER INFORMATION CONTACT: Dr. Frank Gostomski, Criteria and Standards Division (WH-585), U.S. Environmental Protection Agency, 401 M Street SW., Washington, D.C. 20460, (202) 245-3030.

SUPPLEMENTARY INFORMATION

Background

Section 304(a)(1) of the Clean Water Act (33 U.S.C. 1314(a)(1)) requires EPA to publish and periodically update ambient water quality criteria. These criteria are to reflect the latest scientific knowledge on the identifiable effects of pollutants on public health and welfare, aquatic life, and recreation.

EPA has periodically issued ambient water quality criteria, beginning in 1973 with publication of the "Blue Book" (Water Quality Criteria 1972). In 1976, the "Red Book" (Quality Criteria for Water) was published. On November 28, 1980 (45 FR 79318) and February 15, 1984 (49 FR 5831), EPA announced the publication of 65 individual ambient water quality criteria documents for pollutants listed as toxic under section 307(a)(1) of the Clean Water Act.

Today EPA is announcing the availability of nine individual water quality criteria documents which update and revise certain criteria previously published in the "Red Book" and in the 1980 ambient water quality criteria documents. The criteria documents for ammonia and chlorine replace criteria previously published in the 1976 "Red Book." The criteria documents for arsenic, cadmium, chromium, copper, cyanide, lead, and mercury replace the aquatic life criteria previously published in the 1980 ambient water quality criteria documents. Draft criteria documents were made available for public comment on February 7, 1984 (49

FR 4551). These final criteria have been derived after consideration of all comments received.

Dated: July 19, 1985.

Edwin C. Johnson,

Acting Assistant Administrator for Water

Appendix A—Summary of Water Quality Criteria

1. Ammonia

Freshwater Aquatic Life

The procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" indicate that, except possibly where a locally important species is very sensitive, freshwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of ammonia does not exceed the recommended criterion more than once every three years on the average and if the one-hour average concentration does not exceed the recommended criterion more than once every three years on the average.

The recommended exceedence frequency of three years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to ammonia exceeds the criterion. Stressed systems, for example one in which several outfalls occur in a limited area, would be expected to require more time for recovery. The resilience of ecosystems and their ability to recover differ greatly, however, and site-specific criteria may be established if adequate justification is provided.

To protect freshwater aquatic life, the criteria for ammonia (in mg/liter unionized NH_3) are based upon ambient water temperature and pH with one-hour and four-day average concentrations provided. Criterion concentrations for the pH range 6.5 to 9.0 and the temperature range 0 °C to 30 °C are provided in the following tables. Total ammonia concentrations equivalent to each unionized ammonia concentration are also provided in these tables. There is limited data on the effect of temperature on chronic toxicity. EPA will be conducting additional research on the effects of temperature on ammonia toxicity in order to fill perceived data gaps. Because of this uncertainty, additional site-specific information should be developed before these criteria are used in wasteload allocation modelling. For example, the chronic criteria tabulated for non-salmonids at temperatures much below

20 °C are less certain than those tabulated at temperatures near 20 °C. Where the treatment levels needed to meet these criteria below 20 °C may be substantial, use of site-specific criteria is strongly suggested. Development of such criteria should be based upon site-specific toxicity tests.

The use of criteria in designing waste treatment facilities requires the selection of an appropriate wasteload allocation model. Dynamic models are preferred for the application of these criteria. Limited data or other factors may make their use impractical, in which case one should rely on a steady-state model. The Agency recommends the interim use of 1Q5 or 1Q10 for criterion maximum concentration (CMC) design flow and 7Q5 or 7Q10 for the criterion continuous concentration (CCC) design flow in steady-state models for unstressed and stressed systems respectively. The Agency acknowledges that the CCC stream flow averaging period used for steady-state wasteload allocation modelling may be as long as 30 days in situations involving POTWs designed to remove ammonia where limited variability of effluent pollutant concentration and resultant concentrations in receiving waters can be demonstrated.

In cases where low variability can be demonstrated, longer averaging periods for the ammonia CCC (e.g., 30-day averaging periods) would be acceptable because the magnitude and duration of exceedences above the CCC would be sufficiently limited. These matters are discussed in more detail in the Technical Support Document for Water Quality Based Toxics Control (U.S. EPA, 1985).

(1) ONE-HOUR AVERAGE CONCENTRATIONS FOR AMMONIA*

°F	0 °C	5 °C	10 °C	15 °C	20 °C	25 °C	30 °C
A. Salmonids and Other Sensitive Coldwater Species Present							
Unstressed Ammonia (mg/liter NH ₃)							
6.50	0.0091	0.0129	0.0182	0.026	0.038	0.056	0.078
6.75	0.0148	0.021	0.030	0.042	0.059	0.086	0.119
7.00	0.023	0.033	0.046	0.066	0.093	0.135	0.189
7.25	0.034	0.048	0.066	0.095	0.135	0.195	0.273
7.50	0.048	0.064	0.091	0.128	0.181	0.261	0.361
7.75	0.066	0.089	0.113	0.159	0.22	0.32	0.44
8.00	0.088	0.118	0.130	0.184	0.26	0.38	0.51
8.25	0.118	0.162	0.130	0.184	0.26	0.38	0.51
8.50	0.162	0.222	0.130	0.184	0.26	0.38	0.51
8.75	0.222	0.306	0.130	0.184	0.26	0.38	0.51
9.00	0.306	0.418	0.130	0.184	0.26	0.38	0.51
Total Ammonia (mg/liter NH ₃)							
6.50	36	33	31	30	29	28	27
6.75	38	30	29	27	27	26	25
7.00	39	36	29	24	23	22	21
7.25	39	32	29	24	23	22	21
7.50	39	29	26	23	22	21	20
7.75	39	26	23	21	20	19	18
8.00	39	23	21	20	19	18	17
8.25	39	21	20	19	18	17	16
8.50	39	20	19	18	17	16	15
8.75	39	19	18	17	16	15	14
9.00	39	18	17	16	15	14	13
B. Salmonids and Other Sensitive Coldwater Species Absent							
Unstressed Ammonia (mg/liter NH ₃)							
6.50	0.0091	0.0129	0.0182	0.026	0.038	0.056	0.078
6.75	0.0148	0.021	0.030	0.042	0.059	0.086	0.119
7.00	0.023	0.033	0.046	0.066	0.093	0.135	0.189
7.25	0.034	0.048	0.066	0.095	0.135	0.195	0.273
7.50	0.048	0.064	0.091	0.128	0.181	0.261	0.361
7.75	0.066	0.089	0.113	0.159	0.22	0.32	0.44
8.00	0.088	0.118	0.130	0.184	0.26	0.38	0.51
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8.75	0.222	0.306	0.130	0.184	0.26	0.38	0.51
9.00	0.306	0.418	0.130	0.184	0.26	0.38	0.51
Total Ammonia (mg/liter NH ₃)							
6.50	36	33	31	30	29	28	27
6.75	38	30	29	27	27	26	25
7.00	39	36	29	24	23	22	21
7.25	39	32	29	24	23	22	21
7.50	39	29	26	23	22	21	20
7.75	39	26	23	21	20	19	18
8.00	39	23	21	20	19	18	17
8.25	39	21	20	19	18	17	16
8.50	39	20	19	18	17	16	15
8.75	39	19	18	17	16	15	14
9.00	39	18	17	16	15	14	13

*To convert these values to mg/liter N, multiply by 0.822

1Q5
lowest dayflow in four year pe

(2) 4-DAY AVERAGE CONCENTRATIONS FOR AMMONIA¹

pH	0 C	5 C	10 C	15 C	20 C	25 C	30 C
A. Salmonids or Other Sensitive Coldwater Species Present							
Un-stressed Ammonia (mg/liter NH ₃)							
6.50	0.0007	0.0008	0.0013	0.0019	0.0019	0.0019	0.0019
6.75	0.0012	0.0017	0.0023	0.0033	0.0033	0.0033	0.0033
7.00	0.0021	0.0029	0.0042	0.0059	0.0059	0.0059	0.0059
7.25	0.0037	0.0052	0.0074	0.0105	0.0105	0.0105	0.0105
7.50	0.0066	0.0093	0.0132	0.0188	0.0188	0.0188	0.0188
7.75	0.0108	0.0153	0.022	0.031	0.031	0.031	0.031
8.00	0.0128	0.0177	0.025	0.035	0.035	0.035	0.035
8.25	0.0128	0.0177	0.025	0.035	0.035	0.035	0.035
8.50	0.0128	0.0177	0.025	0.035	0.035	0.035	0.035
8.75	0.0128	0.0177	0.025	0.035	0.035	0.035	0.035
9.00	0.0128	0.0177	0.025	0.035	0.035	0.035	0.035
Total Ammonia (mg/liter NH ₃)							
6.50	2.5	2.4	2.2	2.2	1.48	1.04	0.73
6.75	2.5	2.4	2.2	2.2	1.48	1.04	0.73
7.00	2.5	2.4	2.2	2.2	1.48	1.04	0.74
7.25	2.5	2.4	2.2	2.2	1.50	1.04	0.74
7.50	2.5	2.4	2.2	2.2	1.50	1.06	0.74
7.75	2.3	2.2	2.1	2.0	1.40	0.99	0.71
8.00	1.53	1.44	1.37	1.33	0.85	0.66	0.47
8.25	0.87	0.82	0.78	0.76	0.54	0.39	0.28
8.50	0.48	0.47	0.45	0.44	0.32	0.23	0.17
8.75	0.38	0.37	0.36	0.37	0.19	0.15	0.11
9.00	0.18	0.18	0.18	0.18	0.13	0.10	0.08
B. Salmonids and Other Sensitive Coldwater Species Absent							
Un-stressed Ammonia (mg/liter NH ₃)							
6.50	0.0007	0.0008	0.0013	0.0019	0.0028	0.0038	0.0058
6.75	0.0012	0.0017	0.0023	0.0033	0.0047	0.0067	0.0097
7.00	0.0021	0.0029	0.0042	0.0059	0.0083	0.0113	0.0163
7.25	0.0037	0.0052	0.0074	0.0105	0.0146	0.0206	0.0306
7.50	0.0066	0.0093	0.0132	0.0188	0.026	0.036	0.056
7.75	0.0108	0.0153	0.022	0.031	0.043	0.063	0.093
8.00	0.0128	0.0177	0.025	0.035	0.050	0.070	0.100
8.25	0.0128	0.0177	0.025	0.035	0.050	0.070	0.100
8.50	0.0128	0.0177	0.025	0.035	0.050	0.070	0.100
8.75	0.0128	0.0177	0.025	0.035	0.050	0.070	0.100
9.00	0.0128	0.0177	0.025	0.035	0.050	0.070	0.100
Total Ammonia (mg/liter NH ₃)							
6.50	2.5	2.4	2.2	2.2	2.1	1.48	1.03
6.75	2.5	2.4	2.2	2.2	2.1	1.47	1.04
7.00	2.5	2.4	2.2	2.2	2.1	1.47	1.04
7.25	2.5	2.4	2.2	2.2	2.1	1.48	1.06
7.50	2.5	2.4	2.2	2.2	2.1	1.48	1.08
7.75	2.3	2.2	2.1	2.0	1.88	1.36	1.00
8.00	1.53	1.44	1.37	1.33	1.31	0.85	0.67
8.25	0.87	0.82	0.78	0.76	0.76	0.54	0.40
8.50	0.48	0.47	0.45	0.44	0.48	0.32	0.25
8.75	0.38	0.37	0.36	0.37	0.27	0.21	0.16
9.00	0.18	0.18	0.18	0.18	0.17	0.14	0.11

¹ To convert these values to mg/liter N, multiply by 0.833.
² Site-specific criteria development is strongly suggested at temperatures above 20°C because of the limited data available to generate the criteria recommendation, and at temperatures below 20°C because of the limited data and because small changes in the criteria may have significant impact on the level of treatment required in meeting the recommended criteria.

Saltwater Aquatic Life

Data available for saltwater species are insufficient to derive a criterion for saltwater.

2. Arsenic**Freshwater Aquatic Life**

The procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" indicate that, except possibly where a locally important species is very sensitive, freshwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of arsenic(III)

does not exceed 190 µg/l more than once every three years on the average and if the one-hour average concentration does not exceed 360 µg/l more than once every three years on the average.

EPA believes that a measurement such as "acid-soluble" would provide a more scientifically correct basis upon which to establish criteria for metals. The criteria were developed on this basis. However, at this time, no EPA approved methods for such a measurement are available to implement the criteria through the regulatory programs of the Agency and the States. The Agency is considering development and approval of methods for a

measurement such as "acid-soluble". Until available, however, EPA recommends applying the criteria using the total recoverable method. This has two impacts: (1) Certain species of some metals cannot be analyzed directly because the total recoverable method does not distinguish between individual oxidation states, and (2) these criteria may be overly protective when based on the total recoverable method.

The recommended exceedence frequency of three years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to arsenic(III) exceeds the criterion. Stressed systems, for example one in which several outfalls occur in a limited area, would be expected to require more time for recovery. The resilience of ecosystems and their ability to recover differ greatly, however, and site-specific criteria may be established if adequate justification is provided.

Not enough data are available to allow derivation of numerical national water quality criteria for freshwater aquatic life for inorganic arsenic(V) or any organic arsenic compound. Inorganic arsenic(V) is acutely toxic to freshwater aquatic animals at concentrations as low as 850 µg/l, and an acute-chronic ratio of 28 was obtained with the fathead minnow. Arsenic(V) affected freshwater aquatic plants at concentrations as low as 48 µg/l. Monosodium methanearsenate (MSMA) is acutely toxic to aquatic animals at concentrations as low as 1,900 µg/l but no data are available concerning chronic toxicity to animals or toxicity to plants.

The use of criteria in designing waste treatment facilities requires the selection of an appropriate wastewater allocation model. Dynamic models are preferred for the application of these criteria. Limited data or other factors may make their use impractical, in which case one should rely on a steady-state model. The Agency recommends the interim use of 1Q5/1Q10 for criterion maximum concentration (CMC) design flow and 7Q5 or 7Q10 for the criterion continuous concentration (CCC) design flow in steady-state models for unstressed and stressed systems respectively. These matters are discussed in more detail in the Technical Support Document for Water Quality Based Toxics Control (U.S. EPA, 1985).

Saltwater Aquatic Life

The procedures described in the "Guidelines for Deriving Numerical

National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" indicate that, except possibly where a locally important species is very sensitive, saltwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of arsenic(III) does not exceed 36 µg/l more than once every three years on the average and if the one-hour average concentration does not exceed 60 µg/l more than once every three years on the average.

EPA believes that a measurement such as "acid-soluble" would provide a more scientifically correct basis upon which to establish criteria for metals. The criteria were developed on this basis. However, at this time, no EPA approved methods for such a measurement are available to implement the criteria through the regulatory programs of the Agency and the States. The Agency is considering development and approval of methods for a measurement such as "acid-soluble". Until available, however, EPA recommends applying the criteria using the total recoverable method. This has two impacts: (1) Certain species of some metals cannot be analyzed directly because the total recoverable method does not distinguish between individual oxidation states, and (2) these criteria may be overly protective when based on the total recoverable method.

The recommended exceedence frequency of three years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to arsenic(III) exceeds the criterion. Stressed systems, for example one in which several outfalls occur in a limited area, would be expected to require more time for recovery. The resilience of ecosystems and their ability to recover differ greatly, however, and site-specific criteria may be established if adequate justification is provided.

Very few data are available concerning the toxicity of any form of arsenic other than inorganic arsenic(III) to saltwater aquatic life. The available data do show that inorganic arsenic(V) is acutely toxic to saltwater animals at concentrations as low as 2.319 µg/l and affected some saltwater plants at 13 to 56 µg/l. No data are available concerning the chronic toxicity of any form of arsenic other than inorganic arsenic(III) to saltwater aquatic life.

The use of criteria in designing waste treatment facilities requires the selection of an appropriate wasteload allocation model. Dynamic models are preferred for the application of these criteria. Limited data or other factors

may make their use impractical, in which case one should rely on a steady-state model. The Agency recommends the interim use of 1Q5 or 1Q10 for criterion maximum concentration (CMC) design flow and 7Q5 or 7Q10 for the criterion continuous concentration (CCC) design flow in steady-state models for unstressed and stressed systems respectively. These matters are discussed in more detail in the Technical Support Document for Water Quality Based Toxics Control (U.S. EPA, 1985).

3. Cadmium

Freshwater Aquatic Life

The procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" indicate that, except possibly where a locally important species is very sensitive, freshwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration (in µg/l) of cadmium does not exceed the numerical value given by $e^{(0.785(\ln(\text{hardness}) - 3.000))}$ more than once every three years on the average and if the one-hour average concentration (in µg/l) does not exceed the numerical value given by $e^{(1.132(\ln(\text{hardness}) - 3.000))}$ more than once every three years on the average. For example, at hardnesses of 50, 100 and 200 mg/l as CaCO₃ the four-day average concentrations of cadmium are 0.66, 1.1 and 2.0 µg/l, respectively, and the one-hour average concentrations are 1.8, 3.9 and 8.6 µg/l. If brook trout, brown trout, and striped bass are as sensitive as some of the data indicate they might not be protected by this criterion.

EPA believes that a measurement such as "acid-soluble" would provide a more scientifically correct basis upon which to establish criteria for metals. The criteria were developed on this basis. However, at this time, no EPA approved methods for such a measurement are available to implement the criteria through the regulatory programs of the Agency and the States. The Agency is considering development and approval of methods for a measurement such as "acid-soluble". Until available, however, EPA recommends applying the criteria using the total recoverable method. This has two impacts: (1) Certain species of some metals cannot be analyzed directly because the total recoverable method does not distinguish between individual oxidation states, and (2) these criteria may be overly protective when based on the total recoverable method.

The recommended exceedence frequency of three years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to cadmium exceeds the criteria. Stressed systems, for example one in which several outfalls occur in a limited area, would be expected to require more time for recovery. The resilience of ecosystems and their ability to recover differ greatly, however, and site-specific criteria may be established if adequate justification is provided.

The use of criteria in designing waste treatment facilities requires the selection of an appropriate wasteload allocation model. Dynamic models are preferred for the application of these criteria. Limited data or other factors may make their use impractical, in which case one should rely on a steady-state model. The Agency recommends the interim use of 1Q5 or 1Q10 for criterion maximum concentration (CMC) design flow and 7Q5 or 7Q10 for the criterion continuous concentration (CCC) design flow in steady-state models for unstressed and stressed systems respectively. These matters are discussed in more detail in the Technical Support Document for Water Quality Based Toxics Control (U.S. EPA, 1985).

Saltwater Aquatic Life

The procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" indicate that, except possibly where a locally important species is very sensitive, saltwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of cadmium does not exceed 9.3 µg/L more than once every three years on the average and if the one-hour average concentration does not exceed 43 µg/L more than once every three years on the average. The little information that is available concerning the sensitivity of the American lobster to cadmium indicates that this important species might not be protected by this criterion.

EPA believes that a measurement such as "acid-soluble" would provide a more scientifically correct basis upon which to establish criteria for metals. The criteria were developed on this basis. However, at this time, no EPA approved methods for such a measurement are available to implement the criteria through the regulatory programs of the Agency and the States. The Agency is considering development and approval of methods for a

measurement such as "acid-soluble". Until available, however, EPA recommends applying the criteria using the total recoverable method. This has two impacts: (1) Certain species of some metals cannot be analyzed directly because the total recoverable method does not distinguish between individual oxidation states, and (2) these criteria may be overly protective when based on the total recoverable method.

The recommended exceedence frequency of three years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to cadmium exceeds the criteria. Stressed systems, for example one in which several outfalls occur in a limited area, would be expected to require more time for recovery. The resilience of ecosystems and their ability to recover differ greatly, however, and site-specific criteria may be established if adequate justification is provided.

The use of criteria in designing waste treatment facilities requires the selection of an appropriate wasteload allocation model. Dynamic models are preferred for the application of these criteria. Limited data or other factors may make their use impractical, in which case one should rely on a steady-state model. The Agency recommends the interim use of 1Q5 or 1Q10 for criterion maximum concentration (CMC) design flow and 7Q5 or 7Q10 for the criterion continuous concentration (CCC) design flow in steady-state models for unstressed and stressed systems respectively. These matters are discussed in more detail in the Technical Support Document for Water Quality Based Toxic Control (U.S. EPA, 1985).

4. Chlorine

Freshwater Aquatic Life

The procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" indicate that, except possibly where a locally important species is very sensitive, freshwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of total residual chlorine does not exceed 11 µg/l more than once every three years on the average and if the one-hour average concentration does not exceed 19 µg/l more than once every three years on the average.

The recommended exceedence frequency of three years is the Agency's best scientific judgment of the average

amount of time it will take an unstressed system to recover from a pollution event in which exposure to chlorine exceeds the criteria. Stressed systems, for example one in which several outfalls occur in a limited area, would be expected to require more time for recovery. The resilience of ecosystems and their ability to recover differ greatly, however, and site-specific criteria may be established if adequate justification is provided.

The use of criteria in designing waste treatment facilities requires the selection of an appropriate wasteload allocation model. Dynamic models are preferred for the application of these criteria. Limited data or other factors may make their use impractical, in which case one should rely on a steady-state model. The Agency recommends the interim use of 1Q5 or 1Q10 for criterion maximum concentration (CMC) design flow and 7Q5 or 7Q10 for the criterion continuous concentration (CCC) design flow in steady-state models for unstressed and stressed systems respectively. These matters are discussed in more detail in the Technical Support Document for Water Quality Based Toxics Control (U.S. EPA, 1985).

Saltwater Aquatic Life

The procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" indicate that saltwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of chlorine-produced oxidants does not exceed 7.5 µg/l more than once every three years on the average and if the one-hour average concentration does not exceed 13 µg/l more than once every three years on the average.

The recommended exceedence frequency of three years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to chlorine exceeds the criteria. Stressed systems, for example one in which several outfalls occur in a limited area, would be expected to require more time for recovery. The resilience of ecosystems and their ability to recover differ greatly, however, and site-specific criteria may be established if adequate justification is provided.

The use of criteria in designing waste treatment facilities requires the selection of an appropriate wasteload allocation model. Dynamic models are preferred for the application of these criteria. Limited data or other factors

may make their use impractical, in which case one should rely on a steady-state model. The Agency recommends the interim use of 1Q5 or 1Q10 for criterion maximum concentration (CMC) design flow and 7Q5 or 7Q10 for the criterion continuous concentration (CCC) design flow in steady-state models for unstressed and stressed systems respectively. These matters are discussed in more detail in the Technical Support Document for Water Quality Based Toxics Control (U.S. EPA, 1985).

5. Chromium

Freshwater Aquatic Life

Chromium (III). The procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" indicate that, except possibly when a locally important species is very sensitive, freshwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration (in µg/l) of chromium (III) does not exceed the numerical value given by $e^{(2.4129 \ln (\text{hardness}) - 1.562)}$ more than once every three years on the average and if the one-hour average concentration (in µg/l) does not exceed the numerical value given by $e^{(2.4129 \ln (\text{hardness}) - 2.000)}$ more than once every three years on the average. For example, at hardnesses of 50, 100, and 200 mg/l as CaCO₃, the four-day average concentrations of chromium(III) are 120, 210, and 370 µg/l and the one-hour average concentrations are 980, 1,700, and 3,100 µg/l.

Chromium(VI). The procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" indicate that, except possibly where a locally important species is very sensitive, freshwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of chromium(VI) does not exceed 11 µg/l more than once every three years on the average and if the one-hour average concentration does not exceed 16 µg/l more than once every three years on the average.

EPA believes that the measurement such as "acid-soluble" would provide a more scientifically correct basis upon which to establish criteria for metals. The criteria were developed on this basis. However, at this time, no EPA approved methods for such a measurement are available to implement the criteria through the regulatory

programs of the Agency and the States. The Agency is considering development and approval of methods for a measurement such as "acid-soluble". Until available, however, EPA recommends applying the criteria using the total recoverable method. This has two impacts: (1) Certain species of some metals cannot be analyzed directly because the total recoverable method does not distinguish between individual oxidation states, and (2) these criteria may be overly protective when based on the total recoverable method.

The recommended exceedence frequency of three years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to chromium exceeds the criteria. Stressed systems, for example one in which several outfalls occur in a limited area, would be expected to require more time for recovery. The resilience of ecosystems and their ability to recover differ greatly, however, and site-specific criteria may be established if adequate justification is provided.

The use of criteria in designing waste treatment facilities requires the selection of an appropriate wasteload allocation model. Dynamic models are preferred for the application of these criteria. Limited data or other factors may make their use impractical, in which case one should rely on a steady-state model. The Agency recommends the interim use of 1Q5 or 1Q10 for criterion maximum concentration (CMC) design flow and 7Q5 or 7Q10 for the criterion continuous concentration (CCC) design flow in steady-state models for unstressed and stressed systems respectively. These matters are discussed in more detail in the Technical Support Document for Water Quality Based Toxics Control (U.S. EPA, 1985).

Saltwater Aquatic Life

Chromium(III). No saltwater criterion can be derived for chromium(III), but 10,300 µg/l is the EC50 for eastern oyster embryos, whereas 50,400 µg/l did not affect a polychaete worm in a life-cycle test.

Chromium(VI). The procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" indicate that, except possibly where a locally important species is very sensitive, saltwater aquatic organisms should not be affected unacceptably if the four-day average concentration of chromium(VI) does not exceed 50 µg/L more than once every three years on the

average, and if the one-hour average concentration does not exceed 1,100 µg/L more than once every three years on the average. Data suggest that the acute toxicity of chromium(VI) is salinity dependent; therefore, the one-hour average concentration may be underprotective at low salinities.

EPA believes that a measurement such as "acid-soluble" would provide a more scientifically correct basis upon which to establish criteria for metals. The criteria were developed on this basis. However, at this time, no EPA approved methods for such a measurement are available to implement the criteria through the regulatory programs of the Agency and the States. The Agency is considering development and approval of methods for a measurement such as "acid-soluble". Until available, however, EPA recommends applying the criteria using the total recoverable method. This has two impacts: (1) Certain species of some metals cannot be analyzed directly because the total recoverable method does not distinguish between individual oxidation states, and (2) these criteria may be overly protective when based on the total recoverable method.

The recommended exceedence frequency of three years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to chromium exceeds the criteria. Stressed systems, for example one in which several outfalls occur in a limited area, would be expected to require more time for recovery. The resilience of ecosystems and their ability to recover differ greatly, however, and site-specific criteria may be established if adequate justification is provided.

The use of criteria in designing waste treatment facilities requires the selection of an appropriate wasteload allocation model. Dynamic models are preferred for the application of these criteria. Limited data or other factors may make their use impractical, in which case one should rely on a steady-state model. The Agency recommends the interim use of 1Q5 or 1Q10 for criterion maximum concentration (CMC) design flow and 7Q5 or 7Q10 for the criterion continuous concentration (CCC) design flow in steady-state models for unstressed and stressed systems respectively. These matters are discussed in more detail in the Technical Support Document for Water Quality Based Toxics Control (U.S. EPA, 1985).

8. Copper

Freshwater Aquatic Life

The procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" indicate that, except possible where a locally important species in very sensitive, freshwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration (in µg/l) of copper does not exceed the numerical value given by $e^{(0.042)(1 - \ln(\text{hardness})) - 1.425}$ more than once every three years on the average and if the one-hour average concentration (in µg/l) does not exceed the numerical value given by $e^{(0.042)(1 - \ln(\text{hardness})) - 1.425}$ than once every three years on the average. For example, at hardness of 50, 100, and 200 mg/l as CaCO₃, the four-day average concentrations of copper are 6.5, 12, and 21 µg/l respectively, and the one-hour average concentrations are 9.2, 18 and 34 µg/L.

EPA believes that a measurement such as "acid-soluble" would provide a more scientifically correct basis upon which to establish criteria for metals. The criteria were developed on this basis. However, at this time, no EPA approved methods for such a measurement are available to implement the criteria through the regulatory programs of the Agency and the States. The Agency is considering development and approval of methods for a measurement such as "acid-soluble". Until available, however, EPA recommends applying the criteria using the total recoverable method. This has two impacts: (1) Certain species of some metals cannot be analyzed directly because the total recoverable method does not distinguish between individual oxidation states, and (2) these criteria may be overly protective when based on the total recoverable method.

The recommended exceedence frequency of three years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to copper exceeds the criteria. Stressed systems, for example one in which several outfalls occur in a limited area, would be expected to require more time for recovery. The resilience of ecosystems and their ability to recover differ greatly, however, and site-specific criteria may be established if adequate justification is provided.

The use of criteria in designing waste treatment facilities requires the selection of an appropriate wasteload

allocation model. Dynamic models are preferred for the application of these criteria. Limited data or other factors may make their use impractical, in which case one should rely on a steady-state model. The Agency recommends the interim use of 1Q5 or 1Q10 for criterion maximum concentration (CMC) design flow and 7Q5 or 7Q10 for the criterion continuous concentration (CCC) design flow in steady-state models for unstressed and stressed systems respectively. These matters are discussed in more detail in the Technical Support Document for Water Quality Based Toxics Control (U.S. EPA, 1985).

Saltwater Aquatic Life

The procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" indicate that, except possibly where locally important species is very sensitive, saltwater aquatic organisms and their uses should not be affected unacceptably if the one-hour average concentration of copper does not exceed 2.9 µg/l more than once every three years on the average.

EPA believes that a measurement such as "acid-soluble" would provide a more scientifically correct basis upon which to establish criteria for metals. The criteria were developed on this basis. However, at this time, no EPA approved methods for such a measurement are available to implement the criteria through the regulatory programs of the Agency and the States. The Agency is considering development and approval of methods for a measurement such as "acid-soluble". Until available, however, EPA recommends applying the criteria using the total recoverable method. This has two impacts: (1) Certain species of some metals cannot be analyzed directly because the total recoverable method does not distinguish between individual oxidation states, and (2) these criteria may be overly protective when based on the total recoverable method.

The recommended exceedence frequency of three years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to copper exceeds the criterion. Stressed systems, for example one in which several outfalls occur in a limited area, would be expected to require more time for recovery. The resilience of ecosystems and their ability to recover differ greatly, however, and site-specific criteria may be established if adequate justification is provided.

The use of criteria in designing waste treatment facilities requires the selection of an appropriate wasteload allocation model. Dynamic models are preferred for the application of these criteria. Limited data or other factors may make their use impractical, in which case one should rely on a steady-state model. The Agency recommends the interim use of 1Q5 or 1Q10 for criterion maximum concentration (CMC) design flow and 7Q5 or 7Q10 for the criterion continuous concentration (CCC) design flow in steady-state models for unstressed and stressed systems respectively. These matters are discussed in more detail in the Technical Support Document for Water Quality Based Toxics Control (U.S. EPA, 1985).

7. Cyanide

Freshwater Aquatic Life

The procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" indicate that, except possibly where a locally important species is very sensitive, freshwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of cyanide does not exceed 5.2 µg/l more than once every three years on the average and if the one-hour average concentration does not exceed 22 µg/l more than once every three years on the average.

EPA believes that a measure such as free cyanide would provide a more scientifically correct basis upon which to establish criteria for cyanide. The criteria were developed on this basis. However, at this time, no EPA approved methods for free cyanide are available to implement the criteria through the regulatory programs of the Agency and the States. The Agency is considering development and approval of methods for free cyanide. Until available, however, EPA recommends applying the criteria using the total cyanide method. These criteria may be overly protective when based on the total cyanide method.

The recommended exceedence frequency of three years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which cyanide exceeds the criterion. Stressed systems, for example one in which several outfalls occur in a limited area, would be expected to require more time for recovery. The resilience of ecosystems and their ability to recover differ greatly, however, and site-specific

criteria may be established if adequate justification is provided.

The use of criteria in designing waste treatment facilities requires the selection of an appropriate wasteload allocation model. Dynamic models are preferred for the application of these criteria. Limited data or other factors may make their use impractical, in which case one should rely on a steady-state model. The Agency recommends the interim use of 1Q5 or 1Q10 for criterion maximum concentration (CMC) design flow and 7Q5 or 7Q10 for the criterion continuous concentration (CCC) design flow in steady-state models for unstressed and stressed systems respectively. These matters are discussed in more detail in the Technical Support Document for Water Quality Based Toxics Control (U.S. EPA, 1985).

Saltwater Aquatic Life

The procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" indicate that, except possibly where a locally important species is very sensitive, saltwater aquatic organisms and their uses should not be affected unacceptably if the one-hour average concentration of cyanide does not exceed 1.0 µg/L more than once three years on the average.

EPA believes that a measurement such as free cyanide would provide a more scientifically correct basis upon which to establish criteria for cyanide. The criteria were developed on this basis. However, at this time, no EPA approved methods for such a method are available to implement the criteria through the regulatory programs of the Agency and the States. The Agency is considering development and approval of methods for a measurement such as free cyanide. Until available, however, EPA recommends applying the criteria using the total cyanide method. These criteria may be overly protective when based on the total cyanide method.

The recommended exceedence frequency of three years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to cyanide exceeds the criterion. Stressed systems, for example one in which several outfalls occur in a limited area, would be expected to require more time for recovery. The resilience of ecosystems and their ability to recover differ greatly, however, and site-specific criteria may be established if adequate justification is provided.

The use of criteria in designing waste treatment facilities requires the selection of an appropriate wasteload allocation model. Dynamic models are preferred for the application of these criteria. Limited data or other factors may make their use impractical, in which case one should rely on a steady-state model. The Agency recommends the interim use of 1Q5 or 1Q10 for criterion maximum concentration (CMC) design flow and 7Q5 or 7Q10 for the criterion continuous concentration (CCC) design flow in steady-state models for unstressed and stressed systems respectively. These matters are discussed in more detail in the Technical Support Document for Water Quality Based Toxics Control (U.S. EPA, 1985).

8. Lead

Freshwater Aquatic Life

The procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" indicate that, except possibly where a locally important species is very sensitive, freshwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration (in $\mu\text{g/l}$) of lead does not exceed the numerical value given by $e^{(1.25 \ln(\text{hardness}) - 4.55)}$ more than once every three years on the average and if the one-hour average concentration (in $\mu\text{g/l}$) does not exceed the numerical value given by $e^{(1.25 \ln(\text{hardness}) - 1.10)}$ more than once every three years on the average. For example, at hardnesses of 50, 100, and 200 mg/l as CaCO_3 the 4-day average concentrations of lead are 1.3, 3.2, and 7.7 $\mu\text{g/L}$ respectively, and the one-hour average concentrations are 34, 83, and 200 $\mu\text{g/L}$.

EPA believes that a measurement such as "acid-soluble" would provide a more scientifically correct basis upon which to establish criteria for metals. The criteria were developed on this basis. However, at this time, no EPA approved methods for such a measurement are available to implement the criteria through the regulatory programs of the Agency and the States. The Agency is considering development and approval of methods for a measurement such as acid-soluble. Until available, however, EPA recommends applying the criteria using the total recoverable method. This has two impacts: (1) Certain species of some metals cannot be analyzed directly because the total recoverable method cannot distinguish between individual oxidation states, and (2) these criteria

may be overly protective when based on the total recoverable method.

The recommended exceedence frequency of three years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to lead exceeds the criterion. Stressed systems, for example one in which several outfalls occur in a limited area, would be expected to require more time for recovery. The resilience of ecosystems and their ability to recover differ greatly, however, and site-specific criteria may be established if adequate justification is provided.

The use of criteria in designing waste treatment facilities requires the selection of an appropriate wasteload allocation model. Dynamic models are preferred for the application of these criteria. Limited data or other factors may make their use impractical, in which case one should rely on a steady-state model. The Agency recommends the interim use of 1Q5 or 1Q10 for criterion maximum concentration (CMC) design flow and 7Q5 or 7Q10 for the criterion continuous concentration (CCC) design flow in steady-state models for unstressed and stressed systems respectively. These matters are discussed in more detail in the Technical Support Document for Water Quality Based Toxics Control (U.S. EPA, 1985).

Saltwater Aquatic Life

The procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" indicate that, except possibly where a locally important species is very sensitive, saltwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of lead does not exceed 5.6 $\mu\text{g/l}$ more than once every three years on the average and if the one-hour average concentration does not exceed 140 $\mu\text{g/l}$ more than once every three years on the average.

EPA believes that a measurement such as "acid-soluble" would provide a more scientifically correct basis upon which to establish criteria for metals. The criteria were developed on this basis. However, at this time, no EPA approved methods for such a measurement are available to implement the criteria through the regulatory programs of the Agency and the States. The Agency is considering development and approval of methods for a measurement such as acid-soluble. Until available, however, EPA recommends applying the criteria using the total

recoverable method. This has two impacts: (1) Certain species of some metals cannot be analyzed directly because the total recoverable method does not distinguish between individual oxidation states, and (2) these criteria may be overly protective when based on the total recoverable method.

The recommended exceedence frequency of three years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to lead exceeds the criterion. Stressed systems, for example one in which several outfalls occur in a limited area, would be expected to require more time for recovery. The resilience of ecosystems and their ability to recover differ greatly, however, and site-specific criteria may be established if adequate justification is provided.

The use of criteria in designing waste treatment facilities requires the selection of an appropriate wasteload allocation model. Dynamic models are preferred for the application of these criteria. Limited data or other factors may make their use impractical, in which case one should rely on a steady-state model. The Agency recommends the interim use of 1Q5 or 1Q10 for criterion maximum concentration (CMC) design flow and 7Q5 or 7Q10 for the criterion continuous concentration (CCC) design flow in steady-state models for unstressed and stressed systems respectively. These matters are discussed in more detail in the Technical Support Document for Water Quality Based Toxics Control (U.S. EPA, 1985).

9. Mercury

Freshwater Aquatic Life

The procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" indicate that, except possibly where a locally important species is very sensitive, freshwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of mercury does not exceed 0.012 $\mu\text{g/l}$ more than once every three years on the average and if the one-year average concentration does not exceed 2.4 $\mu\text{g/l}$ more than once every three years on the average. If the four-day average concentration exceeds 0.012 $\mu\text{g/l}$ more than once in a three year period, the edible portion of consumed species should be analyzed to determine whether the concentration of

methylmercury exceeds the FDA action level.

EPA believes that a measurement such as "acid-soluble" would provide a more scientifically correct basis upon which to establish criteria for metals. The criteria were developed on this basis. However, at this time, no EPA approved methods for such a measurement are available to implement the criteria through the regulatory programs of the Agency and the States. The Agency is considering development and approval of methods for a measurement such as "acid-soluble". Until available, however, EPA recommends applying the criteria using the total recoverable method. This has two impacts: (1) Certain species of some metals cannot be analyzed directly because the total recoverable method does not distinguish between individual oxidation states, and (2) these criteria may be overly protective when based on the total recoverable method.

The recommended exceedence frequency of three years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to mercury exceeds the criterion. Stressed systems, for example one in which several outfalls occur in a limited area, would be expected to require more time for recovery. The resilience of ecosystems and their ability to recover differ greatly, however, and site-specific criteria may be established if adequate justification is provided.

The use of criteria in designing waste treatment facilities requires the selection of an appropriate wasteload allocation model. Dynamic models are preferred for the application of these criteria. Limited data or other factors may make their use impractical, in which case one should rely on a steady-state model. The Agency recommends the interim use of 1Q5 or 1Q10 for criterion maximum concentration (CMC) design flow and 7Q5 or 7Q10 for the criterion continuous concentration (CCC) design flow in steady-state models for unstressed and stressed systems respectively. These matters are discussed in more detail in the Technical Support Document for Water Quality Based Toxics Control (U.S. EPA, 1985).

Saltwater Aquatic Life

The procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" indicate that, except possibly where a locally important species is very sensitive, saltwater

aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of mercury does not exceed 0.025 µg/l more than once every three years on the average and if the one-hour average concentration does not exceed 2.1 µg/l more than once every three years on the average. If the four-day average concentration exceeds 0.025 µg/l more than once in a three-year period, the edible portion of consumed species should be analyzed to determine whether the concentration of methylmercury exceeds the FDA action level.

EPA believes that a measurement such as "acid-soluble" would provide a more scientifically correct basis upon which to establish criteria for metals. The criteria were developed on this basis. However, at this time, no EPA approved methods for such a measurement are available to implement the criteria through the regulatory programs of the Agency and the States. The Agency is considering development and approval of methods for a measurement such as "acid-soluble". Until available, however, EPA recommends applying the criteria using the total recoverable method. This has two impacts: (1) Certain species of some metals cannot be analyzed directly because the total recoverable method does not distinguish between individual oxidation states, and (2) these criteria may be overly protective when based on the total recoverable method.

The recommended exceedence frequency of three years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to mercury exceeds the criterion. Stressed systems, for example one in which several outfalls occur in a limited area, would be expected to require more time for recovery. The resilience of ecosystems and their ability to recover differ greatly, however, and site-specific criteria may be established if adequate justification is provided.

The use of criteria in designing waste treatment facilities requires the selection of an appropriate wasteload allocation model. Dynamic models are preferred for the application of these criteria. Limited data or other factors may make their use impractical, in which case one should rely on a steady-state model. The Agency recommends the interim use of 1Q5 or 1Q10 for criterion maximum concentration (CMC) design flow and 7Q5 or 7Q10 for the criterion continuous concentration (CCC) design flow in steady-state models for unstressed and stressed

systems respectively. These matters are discussed in more detail in the Technical Support Document for Water Quality Based Toxics Control (U.S. EPA, 1985).

10. Summary of Revisions to "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses"

This revised version of the National Guidelines provides clarifications, additional details, technical and editorial changes from the guidelines published at 45 FR 79341-79347, November 28, 1980. These modifications are the result of comments received on the previous Guidelines and also reflect advances in aquatic toxicology and related fields. The major technical changes are:

1. The acute data required for freshwater animals has been changed to include more tests with invertebrate species.
2. The Final Acute Value is now defined in terms of Genus Mean Acute Values rather than Species Mean Acute Values previously defined. A Genus Mean Acute Value is the geometric mean of all the Species Mean Acute Values available for species in the genus. On the average, species within a genus are toxicologically much more similar than species in different genera, and so the use of Genus Mean Acute Values will prevent data sets from being biased by an overabundance of species in one or a few genera.
3. The Final Acute Value is now calculated using a method that is not subject to the bias encountered with the previous method.
4. The criterion now consists of two numbers—The criterion continuous concentration (CCC) and the criterion maximum concentration (CMC).
 - a. The criterion continuous concentration is now used as a four-day average, rather than as a 24-hour average.
 - b. The criterion maximum concentration is now used as a one hour average, rather than a "not-to-be-exceeded" value.
 - c. Neither of these values should be exceeded more than once every three years on the average.
 - d. Instead of being equal to the Final Acute Value, the criterion maximum concentration is now obtained by dividing the Final Acute Value by 2. The Final Acute Value is intended to protect 95 percent of a group of diverse species, unless an important species is more sensitive. However, a concentration that would severely harm 50 percent of the

fifth percentile or 50 percent of a sensitive important species cannot be considered to be protective of that percentile or that species. Dividing the Final Acute Value by 2 is intended to result in a concentration that will not severely adversely affect too many of the organisms.

5. When available, 96-hour EC50, based on the percentage of organisms immobilized plus the percentage of organisms killed are used instead of 96-hour LC50, for fish; comparable EC50 values are used instead of LC50, for other species.

6. The requirements for using the results of tests with aquatic plants have been made more stringent.

Two appendices (Appendix 1 and 2) were added as part of the guidance. Appendix 1 was added to aid in determining whether a species should be considered resident in North America and its taxonomic classification. Appendix 2 provides guidance for calculating a Final Acute Value.

Appendix B—Response to Comments on "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses"

Introduction—Most "Comments" Listed Below Are Summaries of Individual Comments Which Expressed Similar Points of View

1. Comment—Criteria should only apply outside the mixing zone, not at the end of the pipe or within the mixing zone.

Response—EPA is issuing guidance on mixing zones in the Technical Support Document for Water Quality-based Toxics Control (Office of Water, 1985), hereafter referred to as the TSD. Because one of the two concentrations in each criterion is based on acute toxicity, it might be appropriate to use this concentration from a national or a site-specific criterion when establishing mixing zone standards.

2. Comment—Derivation of water quality criteria for the protection of aquatic organisms and their uses should adequately take into account such things as the precision and accuracy of available methods for measuring concentrations of pollutants, economic and social considerations, etc.

Response—These criteria are intended to be the best scientific judgment of exposures that can be tolerated by aquatic organisms and their uses. Other considerations can be used in the establishment of standards, permits, etc., where permitted by law. The criteria themselves must be based solely on biological, ecological, and

toxicological data concerning the sensitivities of the organisms and their uses. Technological and economic feasibility are not considered in derivation of water quality criteria.

3. Comment—Laboratory data cannot replace experience in the real world.

Response—EPA certainly agrees. Unfortunately few field data are good enough to be useful in deriving national water quality criteria. It is more likely that field data can be used on a site-specific basis to demonstrate whether a criterion, standard, or permit is underprotective. However, in order for field data to be useful, the studies must be designed and executed appropriately. Too many field studies have limited utility because one or more important aspects were not dealt with in a manner that was appropriate to the specific situation of concern.

4. Comment—These Guidelines are incomplete because they do not include protection of human health.

Response—Water quality criteria can be derived to protect a variety of uses and subuses. These Guidelines are only intended to deal with protection of aquatic organisms and their uses. The Guidelines are intended to ensure that the use of aquatic organisms is not subject to restrictions because of exceedence of FDA action levels. Protection of human health is the subject of "Guidelines and Methodology Used in the Preparation of Health Effect Assessment Chapters of the Consent Decree Water Criteria Documents" (U.S. EPA, FR 48: 79347-79357, November 28, 1980).

5. Comment—EPA should provide implementation guidance concerning such things as mixing zones, wastewater allocation, and compliance monitoring.

Response—EPA is providing such guidance in the TSD.

6. Comment—The criteria do not adequately deal with fluctuating concentrations.

Response—EPA has examined the question of fluctuating concentrations and has revised the expression of the criteria to take into account the data that are available concerning relative effects caused by constant and fluctuating concentrations.

7. Comment—The Guidelines do not deal with simultaneous exposure to more than one pollutant.

Response—This is true and is because (a) few useful data are available, (b) the data that are available do not allow the development of useful principles, and (c) there are so many possible combinations of two or more pollutants and each can be present at a variety of concentrations. To deal with such situations, EPA has developed the

toxicity based (whole effluent) approach described in the TSD.

8. Comment—EPA should develop warmwater and coldwater criteria, regional criteria, etc.

Response—EPA knows of no way to geographically subdivide aquatic species so that derivation of criteria for special circumstances would be worth the effort. Community composition changes gradually from area to area. Also, the distinction between warmwater and coldwater species is only reasonably useful because of all the coolwater species. Even the distinction between fresh water and salt water is vague because the waters and their respective fauna mix in upper estuaries. EPA does allow the derivation of site-specific criteria so that pertinent differences between waters, ecosystems, etc., can be appropriately taken into account.

9. Comment—All criteria issued in 1980 should be revised using the new Guidelines.

Response—EPA is selecting pollutants for which criteria should be derived using the new Guidelines. All the pollutants included in the Red Book and in the 1980 criteria documents, as well as other pollutants, are being considered. How many new or revised criteria are derived will depend on the availability of data and resources.

10. Comment—The criteria do not deal with the effects of pH, dissolved oxygen, dissolved solids, temperature, etc., on toxicity and bioaccumulation.

Response—If data are available to demonstrate that a criterion can be quantitatively related to such a factor, then the criterion should be related to that factor. Thus criteria for some metals are hardness-dependent and the criterion for ammonia is both pH and temperature-dependent. All criteria are subject to site-specific modification so that as many factors as desired can be appropriately taken into account.

11. Comment—Criteria do not deal with uptake from food.

Response—This is a potential weakness for all pollutants, but it is particularly of concern for those that are or could be limited by FDA action levels or effects on wildlife predators. When this is a potential problem for an existing discharge, the most pertinent information can be obtained by analyzing edible tissue of appropriate exposed species for the pollutant of concern.

12. Comment—EPA should make available its rationale for selecting pollutants for which criteria are to be derived.

low as 7.350 µg/l and would occur at lower concentrations among species that are more sensitive than those tested. No definitive data are available concerning the chronic toxicity of acrylonitrile to sensitive freshwater aquatic life but mortality occurs at concentrations as low as 2.600 µg/l with a fish species exposed for 30 days.

Saltwater Aquatic Life

Only one saltwater species has been tested with acrylonitrile and no statement can be made concerning acute or chronic toxicity.

Human Health

For the maximum protection of human health from the potential carcinogenic effects due to exposure of acrylonitrile through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero based on the non-threshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels which may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-6} , 10^{-5} , and 10^{-4} . The corresponding criteria are .56 µg/l, .056 µg/l and .0056 µg/l, respectively. If the above estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 6.3 µg/l, .63 µg/l, and .063 µg/l, respectively. Other concentrations representing different risk levels may be calculated by use of the Guidelines. The risk estimate range is presented for information purposes and does not represent an Agency judgment on an "acceptable" risk level.

Aldrin-Dieldrin

Dieldrin

Freshwater Aquatic Life

For dieldrin the criterion to protect fresh water aquatic life as derived using the Guidelines is 0.0019 µg/l as a 24-hour average and the concentration should not exceed 2.5 µg/l at any time.

Saltwater Aquatic Life

For dieldrin the criterion to protect saltwater aquatic life as derived using the Guidelines is 0.0019 µg/l as a 24-hour average and the concentration should not exceed 0.71 µg/l at any time.

Human Health

For the maximum protection of human health from the potential carcinogenic effects due to exposure of dieldrin through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero based on the non-threshold

assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels which may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-6} , 10^{-5} , and 10^{-4} . The corresponding criteria are .71 ng/l, .071 ng/l, and .0071 ng/l, respectively. If the above estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are .76 ng/l, .076 ng/l, and .0076 ng/l respectively. Other concentrations representing different risk levels may be calculated by use of the Guidelines. The risk estimate range is presented for information purposes and does not represent an Agency judgment on an "acceptable" risk level.

Aldrin

Freshwater Aquatic Life

For freshwater aquatic life the concentration of aldrin should not exceed 3.0 µg/l at any time. No data are available concerning the chronic toxicity of aldrin to sensitive freshwater aquatic life.

Saltwater Aquatic Life

For saltwater aquatic life the concentration of aldrin should not exceed 1.3 µg/l at any time. No data are available concerning the chronic toxicity of aldrin to sensitive saltwater aquatic life.

Human Health

For the maximum protection of human health from the potential carcinogenic effects due to exposure of aldrin through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero based on the non-threshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels which may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-6} , 10^{-5} , and 10^{-4} . The corresponding criteria are .74 ng/l, .074 ng/l, and .0074 ng/l, respectively. If the above estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are .79 ng/l, .079 ng/l, and .0079 ng/l, respectively. Other concentrations representing different risk levels may be calculated by use of the Guidelines. The risk estimate range is presented for information purposes and does not represent an Agency judgment on an "acceptable" risk level.

Antimony

Freshwater Aquatic Life

The available data for antimony indicate that acute and chronic toxicity to freshwater aquatic life occur at concentrations as low as 9.000 and 1.600 µg/l, respectively, and would occur at lower concentrations among species that are more sensitive than those tested. Toxicity to algae occurs at concentrations as low as 810 µg/l.

Saltwater Aquatic Life

No saltwater organisms have been adequately tested with antimony, and no statement can be made concerning acute or chronic toxicity.

Human Health

For the protection of human health from the toxic properties of antimony ingested through water and contaminated aquatic organisms, the ambient water criterion is determined to be 146 µg/l.

For the protection of human health from the toxic properties of antimony ingested through contaminated aquatic organisms alone, the ambient water criterion is determined to be 45,000 µg/l.

Arsenic

Freshwater Aquatic Life

For freshwater aquatic life the concentration of total recoverable trivalent inorganic arsenic should not exceed 440 µg/l at any time. Short-term effects on embryos and larvae of aquatic vertebrate species have been shown to occur at concentrations as low as 40 µg/l.

Saltwater Aquatic Life

The available data for total recoverable trivalent inorganic arsenic indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 506 µg/l and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of trivalent inorganic arsenic to sensitive saltwater aquatic life.

Human Health

For the maximum protection of human health from the potential carcinogenic effects due to exposure of arsenic through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero based on the non-threshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels which may result in incremental increase of cancer risk over the lifetime are

by $c(t) = (C_0 \cdot e^{-kt})$ as a 24-hour average and the concentration (in $\mu\text{g/l}$) should not exceed the numerical value given by $c(t) = (C_0 \cdot e^{-kt})$ at any time. For example, a hardnesses of 50, 100, and 200 mg/l as CaCO_3 the criteria are 0.012, 0.023, and 0.081 $\mu\text{g/l}$ respectively, and the concentration of total recoverable cadmium should not exceed 1.5, 3.0 and 6.3 $\mu\text{g/l}$, respectively, at any time.

Saltwater Aquatic Life

For total recoverable cadmium the criterion to protect saltwater aquatic life as derived using the Guidelines is 4.5 $\mu\text{g/l}$ as a 24-hour average and the concentration should not exceed 59 $\mu\text{g/l}$ at any time.

Human Health

The ambient water quality criterion for cadmium is recommended to be identical to the existing drinking water standard which is 10 $\mu\text{g/l}$. Analysis of the toxic effects data resulted in a calculated level which is protective of human health against the ingestion of contaminated water and contaminated aquatic organisms. The calculated value is comparable to the present standard. For this reason a selective criterion based on exposure solely from consumption of 0.3 grams of aquatic organisms was not derived.

Carbon Tetrachloride - U211

Freshwater Aquatic Life

The available data for carbon tetrachloride indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 38,200 $\mu\text{g/l}$ and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of carbon tetrachloride to sensitive freshwater aquatic life.

Saltwater Aquatic Life

The available data for carbon tetrachloride indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 80,000 $\mu\text{g/l}$ and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of carbon tetrachloride to sensitive saltwater aquatic life.

Human Health

For the maximum protection of human health from the potential carcinogenic effects due to exposure of carbon tetrachloride through ingestion of contaminated water and contaminated aquatic organisms the ambient water concentration should be zero based on

the non-threshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels which may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-6} , 10^{-5} , and 10^{-4} . The corresponding criteria are 4.0 $\mu\text{g/l}$, 40 $\mu\text{g/l}$, and 04 $\mu\text{g/l}$, respectively. If the above estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 69.4 $\mu\text{g/l}$, 6.94 $\mu\text{g/l}$, and .69 $\mu\text{g/l}$, respectively. Other concentrations representing different risk levels may be calculated by use of the Guidelines. The risk estimate range is presented for information purposes and does not represent an Agency judgment on an "acceptable" risk level.

Chlordane - U036

Freshwater Aquatic Life

For chlordane the criterion to protect freshwater aquatic life as derived using the Guidelines is 0.0043 $\mu\text{g/l}$ as a 24-hour average and the concentration should not exceed 2.4 $\mu\text{g/l}$ at any time.

Saltwater Aquatic Life

For chlordane the criterion to protect saltwater aquatic life as derived using the Guidelines is 0.0040 $\mu\text{g/l}$ as a 24-hour average and the concentration should not exceed 0.09 $\mu\text{g/l}$ at any time.

Human Health

For the maximum protection of human health from the potential carcinogenic effects due to exposure of chlordane through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero based on the non-threshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels which may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-6} , 10^{-5} , and 10^{-4} . The corresponding criteria are 4.8 ng/l , .48 ng/l , and .048 ng/l , respectively. If the above estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 4.8 ng/l , .48 ng/l , and .048 ng/l , respectively. Other concentrations representing different risk levels may be calculated by use of the Guidelines. The risk estimate range is presented for information purposes and does not represent an Agency judgment on an "acceptable" risk level.

Chlorinated Benzenes

Freshwater Aquatic Life

The available data for chlorinated benzenes indicate that acute toxicity to freshwater aquatic life occurs at

concentrations as low as 230 $\mu\text{g/l}$ and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of the more toxic of the chlorinated benzenes to sensitive freshwater aquatic life but toxicity occurs at concentrations as low as 50 $\mu\text{g/l}$ for a fish species exposed for 7.5 days.

Saltwater Aquatic Life

The available data for chlorinated benzenes indicate that acute and chronic toxicity to saltwater aquatic life occur at concentrations as low as 180 and 129 $\mu\text{g/l}$, respectively, and would occur at lower concentrations among species that are more sensitive than those tested.

Human Health

For the maximum protection of human health from the potential carcinogenic effects due to exposure of hexachlorobenzene through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero based on the non-threshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels which may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-6} , 10^{-5} , and 10^{-4} . The corresponding recommended criteria are 7.2 ng/l , .72 ng/l , and .072 ng/l , respectively. If the above estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 7.4 ng/l , .74 ng/l , and .074 ng/l , respectively.

For the protection of human health from the toxic properties of 1,2,4,5-tetrachlorobenzene ingested through water and contaminated aquatic organisms, the ambient water criterion is determined to be 38 $\mu\text{g/l}$.

For the protection of human health from the toxic properties of 1,2,4,5-tetrachlorobenzene ingested through contaminated aquatic organisms alone, the ambient water criterion is determined to be 48 $\mu\text{g/l}$.

For the protection of human health from the toxic properties of pentachlorobenzene ingested through water and contaminated aquatic organisms, the ambient water criterion is determined to be 74 $\mu\text{g/l}$.

For the protection of human health from the toxic properties of pentachlorobenzene ingested through contaminated aquatic organisms alone, the ambient water criterion is determined to be 85 $\mu\text{g/l}$.

Using the present guidelines, a satisfactory criterion cannot be derived

more sensitive than those tested. No data are available concerning the chronic toxicity of chlorinated naphthalenes to sensitive freshwater aquatic life.

Saltwater Aquatic Life

The available data for chlorinated naphthalenes indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 7.5 µg/l and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of chlorinated naphthalenes to sensitive saltwater aquatic life.

Human Health

Using the present guidelines, a satisfactory criterion cannot be derived at this time due to the insufficiency in the available data for chlorinated naphthalenes.

Chlorinated Phenols

Freshwater Aquatic Life

The available freshwater data for chlorinated phenols indicate that toxicity generally increases with increasing chlorination, and that acute toxicity occurs at concentrations as low as 30 µg/l for 4-chloro-2-methylphenol to greater than 500,000 µg/l for other compounds. Chronic toxicity occurs at concentrations as low as 970 µg/l for 2,4,6-trichlorophenol. Acute and chronic toxicity would occur at lower concentrations among species that are more sensitive than those tested.

Saltwater Aquatic Life

The available saltwater data for chlorinated phenols indicate that toxicity generally increases with increasing chlorination and that acute toxicity occurs at concentrations as low as 440 µg/l for 2,3,5,6-tetrachlorophenol and 29,700 µg/l for 4-chlorophenol. Acute toxicity would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of chlorinated phenols to sensitive saltwater aquatic life.

Human Health

Sufficient data is not available for 3-monochlorophenol to derive a level which would protect against the potential toxicity of this compound. Using available organoleptic data, for controlling undesirable taste and odor quality of ambient water, the estimated level is 0.1 µg/l. It should be recognized that organoleptic data as a basis for establishing a water quality criteria have limitations and have no

demonstrated relationship to potential adverse human health effects.

Sufficient data is not available for 4-monochlorophenol to derive a level which would protect against the potential toxicity of this compound. Using available organoleptic data, for controlling undesirable taste and odor quality of ambient water, the estimated level is 0.1 µg/l. It should be recognized that organoleptic data as a basis for establishing a water quality criteria have limitations and have no demonstrated relationship to potential adverse human health effects.

Sufficient data is not available for 2,3-dichlorophenol to derive a level which would protect against the potential toxicity of this compound. Using available organoleptic data, for controlling undesirable taste and odor quality of ambient water, the estimated level is .04 µg/l. It should be recognized that organoleptic data as a basis for establishing a water quality criteria have limitations and have no demonstrated relationship to potential adverse human health effects.

Sufficient data is not available for 2,5-dichlorophenol to derive a level which would protect against the potential toxicity of this compound. Using available organoleptic data, for controlling undesirable taste and odor quality of ambient water, the estimated level is .5 µg/l. It should be recognized that organoleptic data as a basis for establishing a water quality criteria have limitations and have no demonstrated relationship to potential adverse human health effects.

Sufficient data is not available for 2,6-dichlorophenol to derive a level which would protect against the potential toxicity of this compound. Using available organoleptic data, for controlling undesirable taste and odor quality of ambient water, the estimated level is .2 µg/l. It should be recognized that organoleptic data as a basis for establishing a water quality criteria have limitations and have no demonstrated relationship to potential adverse human health effects.

Sufficient data is not available for 3,4-dichlorophenol to derive a level which would protect against the potential toxicity of this compound. Using available organoleptic data, for controlling undesirable taste and odor quality of ambient water, the estimated level is .3 µg/l. It should be recognized that organoleptic data as a basis for establishing a water quality criteria have limitations and have no demonstrated relationship to potential adverse human health effects.

Sufficient data is not available for 2,3,4,6-tetrachlorophenol to derive a

level which would protect against the potential toxicity of this compound. Using available organoleptic data, for controlling undesirable taste and odor quality of ambient water, the estimated level is 1 µg/l. It should be recognized that organoleptic data as a basis for establishing a water quality criteria have limitations and have no demonstrated relationship to potential adverse human health effects.

For comparison purposes, two approaches were used to derive criterion levels for 2,4,5-trichlorophenol. Based on available toxicity data, for the protection of public health, the derived level is 2.6 mg/l. Using available organoleptic data, for controlling undesirable taste and odor quality of ambient water, the estimated level is 1.0 µg/l. It should be recognized that organoleptic data as a basis for establishing a water quality criteria have limitations and have no demonstrated relationship to potential adverse human health effects.

For the maximum protection of human health from the potential carcinogenic effects due to exposure of 2,4,6-trichlorophenol through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero based on the non-threshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels which may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-6} , 10^{-5} , and 10^{-4} . The corresponding criteria are 12 µg/l, 1.2 µg/l, and .12 µg/l respectively. If the above estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 36 µg/l, 3.6 µg/l, and .36 µg/l, respectively. Other concentrations representing different risk levels may be calculated by use of the Guidelines. The risk estimate range is presented for information purposes and does not represent an Agency judgment on an "acceptable" risk level.

Using available organoleptic data, for controlling undesirable taste and odor quality of ambient water, the estimated level is 2 µg/l. It should be recognized that organoleptic data as a basis for establishing a water quality criterion have limitations and have no demonstrated relationship to potential adverse human health effects.

Sufficient data is not available for 2-methyl-4-chlorophenol to derive a level which would protect against any potential toxicity of this compound. Using available organoleptic data, for controlling undesirable taste and odor quality of ambient water, the estimated level is 1800 µg/l. It should be

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demonstrated relationship to potential adverse human health effects.

Chromium^{VI} - 855

Freshwater Aquatic Life

For total recoverable hexavalent chromium the criterion to protect freshwater aquatic life as derived using the Guidelines is 0.29 µg/l as a 24-hour average and the concentration should not exceed 21 µg/l at any time.

For freshwater aquatic life the concentration (in µg/l) of total recoverable trivalent chromium should not exceed the numerical value given by $e(1.06(\ln(\text{hardness})) + 3.48)$ at any time. For example, at hardnesses of 50, 100 and 200 mg/l as CaCO₃, the concentration of total recoverable trivalent chromium should not exceed 2,200, 4,700, and 9,900 µg/l, respectively, at any time. The available data indicate that chronic toxicity to freshwater aquatic life occurs at concentrations as low as 44 µg/l and would occur at lower concentrations among species that are more sensitive than those tested.

Saltwater Aquatic Life

For total recoverable hexavalent chromium the criterion to protect saltwater aquatic life as derived using the Guidelines is 18 µg/l as a 24-hour average and the concentration should not exceed 1,200 µg/l at any time.

For total recoverable trivalent chromium, the available data indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 10,300 µg/l, and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of trivalent chromium to sensitive saltwater aquatic life.

Human Health

For the protection of human health from the toxic properties of Chromium III ingested through water and contaminated aquatic organisms, the ambient water criterion is determined to be 170 mg/l.

For the protection of human health from the toxic properties of Chromium III ingested through contaminated aquatic organisms alone, the ambient water criterion is determined to be 3433 mg/l.

The ambient water quality criterion for total Chromium VI is recommended to be identical to the existing drinking water standard which is 30 µg/l. Analysis of the toxic effects data resulted in a calculated level which is protective of human health against the ingestion of contaminated water and contaminated aquatic organisms. The

calculated value is comparable to the present standard. For this reason a selective criterion based on exposure solely from consumption of 6.5 grams of aquatic organisms was not derived.

Copper

Freshwater Aquatic Life

For total recoverable copper the criterion to protect freshwater aquatic life as derived using the Guidelines is 3.8 µg/l as a 24-hour average and the concentration (in µg/l) should not exceed the numerical value given by $e(0.94(\ln(\text{hardness})) - 1.23)$ at any time. For example, at hardnesses of 50, 100, and 200 mg/l CaCO₃, the concentration of total recoverable copper should not exceed 12, 22, and 43 µg/l at any time.

Saltwater Aquatic Life

For total recoverable copper the criterion to protect saltwater aquatic life as derived using the Guidelines is 4.0 µg/l as a 24-hour average and the concentration should not exceed 23 µg/l at any time.

Human Health

Sufficient data is not available for copper to derive a level which would protect against the potential toxicity of this compound. Using available organoleptic data, for controlling undesirable taste and odor quality of ambient water, the estimated level is 1 mg/l. It should be recognized that organoleptic data as a basis for establishing a water quality criteria have limitations and have no demonstrated relationship to potential adverse human health effects.

Cyanide - 8030

Freshwater Aquatic Life

For free cyanide (sum of cyanide present as HCN and CN⁻, expressed as CN) the criterion to protect freshwater aquatic life as derived using the Guidelines is 3.5 µg/l as a 24-hour average and the concentration should not exceed 32 µg/l at any time.

Saltwater Aquatic Life

The available data for free cyanide (sum of cyanide present as HCN and CN⁻, expressed as CN) indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 30 µg/l and would occur at lower concentrations among species that are more sensitive than those tested. If the acute-chronic ratio for saltwater organisms is similar to that for freshwater organisms, chronic toxicity would occur at concentrations as low as 2.0 µg/l for the tested species and at lower concentrations among species

that are more sensitive than those tested.

Human Health

The ambient water quality criterion for cyanide is recommended to be identical to the existing drinking water standard which is 200 µg/l. Analysis of the toxic effects data resulted in a calculated level which is protective of human health against the ingestion of contaminated water and contaminated aquatic organisms. The calculated value is comparable to the present standard. For this reason a selective criterion based on exposure solely from consumption of 6.5 grams of aquatic organisms was not derived.

DDT and Metabolites

Freshwater Aquatic Life

DDT - 4061

For DDT and its metabolites the criterion to protect freshwater aquatic life as derived using the Guidelines is 0.0010 µg/l as a 24-hour average and the concentration should not exceed 1.1 µg/l at any time.

TDE

The available data for TDE indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 0.6 µg/l and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of TDE to sensitive freshwater aquatic life.

DDE

The available data for DDE indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 1,050 µg/l and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of DDE to sensitive freshwater aquatic life.

Saltwater Aquatic Life

DDT

For DDT and its metabolites the criterion to protect saltwater aquatic life as derived using the Guidelines is 0.0010 µg/l as a 24-hour average and the concentration should not exceed 0.13 µg/l at any time.

TDE

The available data for TDE indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 3.8 µg/l and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the

Based on available toxicity data, for the protection of public health, the derived level is 3.00 mg/L. Using available organoleptic data, for controlling undesirable taste and odor quality of ambient water, the estimated level is 0.3 µg/L. It should be recognized that organoleptic data as a basis for establishing a water quality criteria have limitations and have no demonstrated relationship to potential adverse human health effects.

Dichloropropanes/Dichloropropenes

Freshwater Aquatic Life 11054

The available data for dichloropropanes indicate that acute and chronic toxicity to freshwater aquatic life occurs at concentrations as low as 23,000 and 3,700 µg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested.

The available data for dichloropropenes indicate that acute and chronic toxicity to freshwater aquatic life occurs at concentrations as low as 6,080 and 244 µg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested.

Saltwater Aquatic Life

The available data for dichloropropanes indicate that acute and chronic toxicity to saltwater aquatic life occurs at concentrations as low as 10,300 and 3,040 µg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested.

The available data for dichloropropenes indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 760 µg/L, and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of dichloropropenes to sensitive saltwater aquatic life.

Human Health

Using the present guidelines, a satisfactory criterion cannot be derived at this time due to the insufficiency in the available data for dichloropropanes.

For the protection of human health from the toxic properties of dichloropropanes ingested through water and contaminated aquatic organisms, the ambient water criterion is determined to be 67 µg/L.

For the protection of human health from the toxic properties of dichloropropenes ingested through contaminated aquatic organisms alone,

the ambient water criterion is determined to be 14.1 mg/L.

2,4-Dimethylphenol

Freshwater Aquatic Life

The available data for 2,4-dimethylphenol indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 2,120 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of dimethylphenol to sensitive freshwater aquatic life.

Saltwater Aquatic Life

No saltwater organisms have been tested with 2,4-dimethylphenol and no statement can be made concerning acute and chronic toxicity.

Human Health

Sufficient data are not available for 2,4-dimethylphenol to derive a level which would protect against the potential toxicity of this compound. Using available organoleptic data, for controlling undesirable taste and odor quality of ambient water, the estimated level is 400 µg/L. It should be recognized that organoleptic data as a basis for establishing a water quality criteria have limitations and have no demonstrated relationship to potential adverse human health effects.

2,4-Dinitrotoluenes

Freshwater Aquatic Life

The available data for 2,4-dinitrotoluenes indicate that acute and chronic toxicity to freshwater aquatic life occurs at concentrations as low as 330 and 230 µg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested.

Saltwater Aquatic Life

The available data for 2,4-dinitrotoluenes indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 590 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of 2,4-dinitrotoluenes to sensitive saltwater aquatic life but a decrease in algal cell numbers occurs at concentrations as low as 370 µg/L.

Human Health

For the maximum protection of human health from the potential carcinogenic effects due to exposure of 2,4-dinitrotoluene through ingestion of contaminated water and contaminated

aquatic organisms, the ambient water concentration should be zero based on the non-threshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels which may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-6} , 10^{-5} , and 10^{-4} . The corresponding criteria are 1.1 µg/L, 0.11 µg/L, and 0.011 µg/L, respectively. If the above estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 91 µg/L, 9.1 µg/L, and 0.91 µg/L, respectively. Other concentrations representing different risk levels may be calculated by use of the Guidelines. The risk estimate range is presented for information purposes and does not represent an Agency judgment on an "acceptable" risk level.

1,2-Diphenylhydrazine

Freshwater Aquatic Life 11059

The available data for 1,2-diphenylhydrazine indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 270 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of 1,2-diphenylhydrazine to sensitive freshwater aquatic life.

Saltwater Aquatic Life

No saltwater organisms have been tested with 1,2-diphenylhydrazine and no statement can be made concerning acute and chronic toxicity.

Human Health

For the maximum protection of human health from the potential carcinogenic effects due to exposure of 1,2-diphenylhydrazine through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero based on the non-threshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels which may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-6} , 10^{-5} , and 10^{-4} . The corresponding criteria are 422 ng/L, 42 ng/L, and 4 ng/L, respectively. If the above estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 3.6 µg/L, 0.36 µg/L, and 0.036 µg/L, respectively. Other concentrations representing different risk levels may be calculated by use of the Guidelines. The risk estimate range is presented for information purposes and does not

1.9 µg/l, 0.19 µg/l, and 0.019 µg/l respectively. If the above estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 157 µg/l, 15.7 µg/l, and 1.57 µg/l, respectively. Other concentrations representing different risk levels may be calculated by use of the Guidelines. The risk estimate range is presented for information purposes and does not represent an Agency judgment on an "acceptable" risk level.

Heptachlor — 9059

Freshwater Aquatic Life

For heptachlor the criterion to protect freshwater aquatic life as derived using the Guidelines is 0.0036 µg/l as a 24-hour average and the concentration should not exceed 0.52 µg/l at any time.

Saltwater Aquatic Life

For heptachlor the criterion to protect saltwater aquatic life as derived using the Guidelines is 0.0036 µg/l as a 24-hour average and the concentration should not exceed 0.063 µg/l at any time.

Human Health

For the maximum protection of human health from the potential carcinogenic effects due to exposure of heptachlor through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero based on the non-threshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels which may result in incremental increase of cancer risk, over the lifetimes are estimated at 10^{-6} , 10^{-5} , and 10^{-4} . The corresponding criteria are 2.76 ng/l, .28 ng/l, and .028 ng/l, respectively. If the above estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 2.85 ng/l, .29 ng/l, and .029 ng/l, respectively. Other concentrations representing different risk levels may be calculated by use of the Guidelines. The risk estimate range is presented for information purposes and does not represent an Agency judgment on an "acceptable" risk level.

Hexachlorobutadiene — 4123

Freshwater Aquatic Life

The available data for hexachlorobutadiene indicate that acute and chronic toxicity to freshwater aquatic life occur at concentrations as low as 90 and 9.3 µg/l, respectively, and would occur at lower concentrations among species that are more sensitive than those tested.

Saltwater Aquatic Life

The available data for hexachlorobutadiene indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 32 µg/l and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of hexachlorobutadiene to sensitive saltwater aquatic life.

Human Health

For the maximum protection of human health from the potential carcinogenic effects due to exposure of hexachlorobutadiene through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero based on the non-threshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels which may result in incremental increase of cancer risk, over the lifetimes are estimated at 10^{-6} , 10^{-5} , and 10^{-4} . The corresponding criteria are 4.47 µg/l, 0.45 µg/l, and 0.045 µg/l, respectively. If the above estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 500 µg/l, 50 µg/l, and 5 µg/l, respectively. Other concentrations representing different risk levels may be calculated by use of the Guidelines. The risk estimate range is presented for information purposes and does not represent an Agency judgment on an "acceptable" risk level.

Hexachlorocyclohexane — 3171

Lindane

Freshwater Aquatic Life

For Lindane the criterion to protect freshwater aquatic life as derived using the Guidelines is 0.080 µg/l as a 24-hour average and the concentration should not exceed 2.0 µg/l at any time.

Saltwater Aquatic Life

For saltwater aquatic life the concentration of lindane should not exceed 0.16 µg/l at any time. No data are available concerning the chronic toxicity of lindane to sensitive saltwater aquatic life.

BHC

Freshwater Aquatic Life

The available data for a mixture of isomers of BHC indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 100 µg/l and would occur at lower concentrations among species that are more sensitive than those tested. No data are available

concerning the chronic toxicity of a mixture of isomers of BHC to sensitive freshwater aquatic life.

Saltwater Aquatic Life

The available data for a mixture of isomers of BHC indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 0.34 µg/l and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of a mixture of isomers of BHC to sensitive saltwater aquatic life.

Human Health

For the maximum protection of human health from the potential carcinogenic effects due to exposure of alpha-HCH through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero based on the non-threshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels which may result in incremental increase of cancer risk, over the lifetimes are estimated at 10^{-6} , 10^{-5} , and 10^{-4} . The corresponding criteria are 42 ng/l, 9.2 ng/l, and .92 ng/l, respectively. If the above estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 310 ng/l, 31.0 ng/l, and 3.1 ng/l, respectively. Other concentrations representing different risk levels may be calculated by use of the Guidelines. The risk estimate range is presented for information purposes and does not represent an Agency judgment on an "acceptable" risk level.

For the maximum protection of human health from the potential carcinogenic effects due to exposure of beta-HCH through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero based on the non-threshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels which may result in incremental increase of cancer risk, over the lifetimes are estimated at 10^{-6} , 10^{-5} , and 10^{-4} . The corresponding criteria are 163 ng/l, 16.3 ng/l, and 1.63 ng/l, respectively. If the above estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 547 ng/l, 54.7 ng/l, and 5.47 ng/l, respectively. Other concentrations representing different risk levels may be calculated by use of the Guidelines. The risk estimate range is presented for information purposes and does not

ingested through water and contaminated aquatic organisms, the ambient water criterion is determined to be 144 ng/l.

For the protection of human health from the toxic properties of mercury ingested through contaminated aquatic organisms alone, the ambient water criterion is determined to be 146 ng/l.

Note.—These values include the consumption of freshwater, estuarine, and marine species.

Naphthalene

Freshwater Aquatic Life

The available data to naphthalene indicate that acute and chronic toxicity to freshwater aquatic life occur at concentrations as low as 2,300 and 620 µg/L respectively, and would occur at lower concentrations among species that are more sensitive than those tested.

Saltwater Aquatic Life

The available data for naphthalene indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 2,350 µg/l and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of naphthalene to sensitive saltwater aquatic life.

Human Health

Using the present guidelines, a satisfactory criterion cannot be derived at this time due to the insufficiency in the available data for naphthalene.

Nickel

Freshwater Aquatic Life

For total recoverable nickel the criterion (in µg/l) to protect freshwater aquatic life as derived using the Guidelines is the numerical value given by $e(0.78[\ln(\text{hardness})] + 1.08)$ as a 24-hour average and the concentration (in µg/l) should not exceed the numerical value given by $e(0.78[\ln(\text{hardness})] + 4.02)$ at any time. For example, at hardnesses of 50, 100, and 200 mg/l as CaCO₃, the criteria are 56, 98, and 160 µg/l, respectively, as 24-hour averages, and the concentrations should not exceed 1,100, 1,800, and 3,100 µg/l, respectively, at any time.

Saltwater Aquatic Life

For total recoverable nickel the criterion to protect saltwater aquatic life as derived using the Guidelines is 7.1 µg/l as a 24-hour average and the concentration should not exceed 140 µg/l at any time.

Human Health

For the protection of human health from the toxic properties of nickel ingested through water and contaminated aquatic organisms, the ambient water criterion is determined to be 13.4 µg/l.

For the protection of human health from the toxic properties of nickel ingested through contaminated aquatic organisms alone, the ambient water criterion is determined to be 100 µg/l.

Nitrobenzene

Freshwater Aquatic Life

The available data for nitrobenzene indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 27,000 µg/l and would occur at lower concentrations among species that are more sensitive than those tested. No definitive data are available concerning the chronic toxicity of nitrobenzene to sensitive freshwater aquatic life.

Saltwater Aquatic Life

The available data for nitrobenzene indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 6,680 µg/l and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of nitrobenzene to sensitive saltwater aquatic life.

Human Health

For comparison purposes, two approaches were used to derive criterion levels for nitrobenzene. Based on available toxicity data, for the protection of public health, the derived level is 19.8 mg/l. Using available organoleptic data, for controlling undesirable taste and odor-quality of ambient water, the estimated level is 30 µg/l. It should be recognized that organoleptic data as a basis for establishing a water quality criteria have limitations and have no demonstrated relationship to potential adverse human health effects.

Nitrophenols

Freshwater Aquatic Life

The available data for nitrophenols indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 230 µg/l and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of nitrophenols to sensitive freshwater aquatic life but toxicity to one species of algae occurs at concentrations as low as 150 µg/l.

Saltwater Aquatic Life

The available data for nitrophenols indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 4,850 µg/l and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of nitrophenols to sensitive saltwater aquatic life.

Human Health

For the protection of human health from the toxic properties of 2,4-dinitro-cresol ingested through water and contaminated aquatic organisms, the ambient water criterion is determined to be 13.4 µg/l.

For the protection of human health from the toxic properties of 2,4-dinitro-cresol ingested through contaminated aquatic organisms alone, the ambient water criterion is determined to be 765 µg/l.

For the protection of human health from the toxic properties of dinitrophenol ingested through water and contaminated aquatic organisms, the ambient water criterion is determined to be 70 µg/l.

For the protection of human health from the toxic properties of dinitrophenol ingested through contaminated aquatic organisms alone, the ambient water criterion is determined to be 14.3 mg/l.

Using the present guidelines, a satisfactory criterion cannot be derived at this time due to the insufficiency in the available data for mononitrophenol.

Using the present guidelines, a satisfactory criterion cannot be derived at this time due to the insufficiency in the available data for tri-nitrophenol.

Nitrosamines

Freshwater Aquatic Life

The available data for nitrosamines indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 5,850 µg/l and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of nitrosamines to sensitive freshwater aquatic life.

Saltwater Aquatic Life

The available data for nitrosamines indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 3,300,000 µg/l and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of nitrosamines to sensitive saltwater aquatic life.

undesirable taste and odor quality of ambient water, the estimated level is 0.3 mg/l. It should be recognized that organoleptic data as a basis for establishing a water quality criterion have limitations and have no demonstrated relationship to potential adverse human health effects.

Phthalate Esters

Freshwater Aquatic Life

The available data for phthalate esters indicate that acute and chronic toxicity to freshwater aquatic life occur at concentrations as low as 940 and 3 µg/l, respectively, and would occur at lower concentrations among species that are more sensitive than those tested.

Saltwater Aquatic Life

The available data for phthalate esters indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 2944 µg/l and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of phthalate esters to sensitive saltwater aquatic life but toxicity to one species of algae occurs at concentrations as low as 3.4 µg/l.

Human Health

For the protection of human health from the toxic properties of dimethyl-phthalate ingested through water and contaminated aquatic organisms, the ambient water criterion is determined to be 313 mg/l.

For the protection of human health from the toxic properties of diethyl-phthalate ingested through contaminated aquatic organisms alone, the ambient water criterion is determined to be 2.9 g/l.

For the protection of human health from the toxic properties of diethyl-phthalate ingested through water and contaminated aquatic organisms, the ambient water criterion is determined to be 330 mg/l.

For the protection of human health from the toxic properties of diethyl-phthalate ingested through contaminated aquatic organisms alone, the ambient water criterion is determined to be 1.8 g/l.

For the protection of human health from the toxic properties of dibutyl-phthalate ingested through water and contaminated aquatic organisms, the ambient water criterion is determined to be 34 mg/l.

For the protection of human health from the toxic properties of dibutyl-phthalate ingested through

contaminated aquatic organisms alone, the ambient water criterion is determined to be 154 mg/l.

For the protection of human health from the toxic properties of di-2-ethylhexyl-phthalate ingested through water and contaminated aquatic organisms, the ambient water criterion is determined to be 15 mg/l.

For the protection of human health from the toxic properties of di-2-ethylhexyl-phthalate ingested through contaminated aquatic organisms alone, the ambient water criterion is determined to be 50 mg/l.

Polychlorinated Biphenyls

Freshwater Aquatic Life

For polychlorinated biphenyls the criterion to protect freshwater aquatic life as derived using the Guidelines is 0.014 µg/l as a 24-hour average. The available data indicate that acute toxicity to freshwater aquatic life probably will only occur at concentrations above 2.0 µg/l and that the 24-hour average should provide adequate protection against acute toxicity.

Saltwater Aquatic Life

For polychlorinated biphenyls the criterion to protect saltwater aquatic life as derived using the Guidelines is 0.030 µg/l as a 24-hour average. The available data indicate that acute toxicity to saltwater aquatic life probably will only occur at concentrations above 10 µg/l and that the 24-hour average should provide adequate protection against acute toxicity.

Human Health

For the maximum protection of human health from the potential carcinogenic effects due to exposure of PCBs through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero based on the non-threshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels which may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-6} , 10^{-5} , and 10^{-4} . The corresponding criteria are .79 ng/l, 0.79 ng/l, and .0079 ng/l, respectively. If the above estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are .79 ng/l, .079 ng/l, and .0079 ng/l, respectively. Other concentrations representing different risk levels may be calculated by use of the Guidelines. The risk estimate range is presented for information purposes and does not

represent an Agency judgment on an "acceptable" risk level.

Polynuclear Aromatic Hydrocarbons (PAHs)

Freshwater Aquatic Life

The limited freshwater data base available for polynuclear aromatic hydrocarbons, mostly from short-term bioconcentration studies with two compounds, does not permit a statement concerning acute or chronic toxicity.

Saltwater Aquatic Life

The available data for polynuclear aromatic hydrocarbons indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 300 µg/l and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of polynuclear aromatic hydrocarbons to sensitive saltwater aquatic life.

Human Health

For the maximum protection of human health from the potential carcinogenic effects due to exposure of PAHs through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero based on the non-threshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels which may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-6} , 10^{-5} , and 10^{-4} . The corresponding criteria are 28 ng/l, 2.8 ng/l, and .28 ng/l, respectively. If the above estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 311 ng/l, 31.1 ng/l, and 3.11 ng/l, respectively. Other concentrations representing different risk levels may be calculated by use of the Guidelines. The risk estimate range is presented for information purposes and does not represent an Agency judgment on an "acceptable" risk level.

Selenium

Freshwater Aquatic Life

For total recoverable inorganic selenite the criterion to protect freshwater aquatic life as derived using the Guidelines is 35 µg/l as a 24-hour average and the concentration should not exceed 280 µg/l at any time.

The available data for inorganic selenate indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 760 µg/l and would occur at lower concentrations among species that are more sensitive

are available concerning the chronic toxicity of toxaphene to sensitive saltwater aquatic life.

Human Health

For the maximum protection of human health from the potential carcinogenic effects due to exposure of toxaphene through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero based on the non-threshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels which may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-6} , 10^{-5} , and 10^{-4} . The corresponding criteria are 7.1 ng/L, .71 ng/L, and .07 ng/L, respectively. If the above estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 7.3 ng/L, .73 ng/L, and .07 ng/L, respectively. Other concentrations representing different risk levels may be calculated by use of the Guidelines. The risk estimate range is presented for information purposes and does not represent an Agency judgment on an "acceptable" risk level.

Trichloroethylene — U223

Freshwater Aquatic Life

The available data for trichloroethylene indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 45,000 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of trichloroethylene to sensitive freshwater aquatic life but adverse behavioral effects occur to one species at concentrations as low as 21,900 µg/L.

Saltwater Aquatic Life

The available data for trichloroethylene indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 2,000 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of trichloroethylene to sensitive saltwater aquatic life.

Human Health

For the maximum protection of human health from the potential carcinogenic effects due to exposure of trichloroethylene through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero based on

the non-threshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels which may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-6} , 10^{-5} , and 10^{-4} . The corresponding criteria are 27 µg/L, 2.7 µg/L, and .27 µg/L, respectively. If the above estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 807 µg/L, 80.7 µg/L, and 8.07 µg/L, respectively. Other concentrations representing different risk levels may be calculated by use of the Guidelines. The risk estimate range is presented for information purposes and does not represent an Agency judgment on an "acceptable" risk level.

Vinyl Chloride — U043

Freshwater Aquatic Life

No freshwater organisms have been tested with vinyl chloride and no statement can be made concerning acute or chronic toxicity.

Saltwater Aquatic Life

No saltwater organisms have been tested with vinyl chloride and no statement can be made concerning acute or chronic toxicity.

Human Health

For the maximum protection of human health from the potential carcinogenic effects due to exposure of vinyl chloride through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero based on the non-threshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels which may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-6} , 10^{-5} , and 10^{-4} . The corresponding criteria are 20 µg/L, 2.0 µg/L, and .2 µg/L, respectively. If the above estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 5,246 µg/L, 525 µg/L, and 52.5 µg/L, respectively. Other concentrations representing different risk levels may be calculated by use of the Guidelines. The risk estimate range is presented for information purposes and does not represent an Agency judgment on an "acceptable" risk level.

Zinc

Freshwater Aquatic Life

For total recoverable zinc the criterion to protect freshwater aquatic life as derived using the Guidelines is 47 µg/L as a 24-hour average and the concentration (in µg/L) should not

exceed the numerical value given by $g(0.001a(\text{hardness})) + 1.00$ at any time. For example, at hardnesses of 50, 100, and 200 mg/L as CaCO₃, the concentration of total recoverable zinc should not exceed 180, 320, and 570 µg/L at any time.

Saltwater Aquatic Life

For total recoverable zinc the criterion to protect saltwater aquatic life as derived using the Guidelines is 58 µg/L as a 24-hour average and the concentration should not exceed 170 µg/L at any time.

Human Health

Sufficient data is not available for zinc to derive a level which would protect against the potential toxicity of this compound. Using available organoleptic data, for controlling undesirable taste and odor quality of ambient water, the estimated level is 5 mg/L. It should be recognized that organoleptic data as a basis for establishing a water quality criteria have limitations and have not demonstrated relationship to potential adverse human health effects.

Appendix B—Guidelines for Deriving Water Quality Criteria for the Protection of Aquatic Life and Its Uses

Introduction

This version of the Guidelines provides clarifications, additional details, and technical and editorial changes in the last version published in the Federal Register [44 FR 15970 (March 13, 1979)]. This version incorporates changes resulting from comments on previous versions and from experience gained during U.S. EPA's use of the previous versions. Future versions of the Guidelines will incorporate new ideas and data as their usefulness is demonstrated.

Criteria may be expressed in several forms. The numerical form is commonly used, but descriptive and procedural forms can be used if numerical criteria are not possible or desirable. The purpose of these Guidelines is to describe an objective, internally consistent and appropriate way of deriving numerical water quality criteria for the protection of the uses of, as well as the presence of, aquatic organisms.

A numerical criterion might be thought of as an estimate of the highest concentration of a substance in water which does not present a significant risk to the aquatic organisms in the water and their uses. Thus the Guidelines are intended to derive criteria which will protect aquatic communities by protecting most of the species and their uses most of the time, but not

(OW-FRL 2522-2)

Water Quality Criteria; Availability of Document**AGENCY:** Environmental Protection Agency.**ACTION:** Notice of Final Ambient Water Quality Criteria Document.

SUMMARY: EPA announces the availability and provides a summary of a final ambient water quality criteria document for 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD). These criteria are published pursuant to section 304(a)(1) of the Clean Water Act.

Availability of Document

A summary of the criteria from this document is published below. Copies of the complete document may be obtained from the person listed below. This document is also available for public inspection and copying during normal business hours at the Public Information Reference Unit, U.S. Environmental Protection Agency, Room 2404 (rear), 401 M St., S.W., Washington, D.C. 20460. As provided in 40 CFR Part 2, a reasonable fee may be charged for copying services. Copies of this document are also available for review in the EPA Regional Office libraries.

FOR FURTHER INFORMATION CONTACT: Dr. Frank Gostomski, Criteria and Standards Division (WH-385), United States Environmental Protection Agency, 401 M Street, S.W., Washington, D.C. 20460, (202) 245-3030.

SUPPLEMENTARY INFORMATION:**Background**

Pursuant to section 304(a)(1) of the Clean Water Act, 33 U.S.C. 1314(a)(1), EPA is required to periodically review and publish criteria for water quality accurately reflecting the latest scientific knowledge:

(A) on the kind and extent of all identifiable effects on health and welfare including, but not limited to, plankton, fish, shellfish, wildlife, plant life, shorelines, beaches, aesthetics, and recreation which may be expected from the presence of pollutants in any body of water, including groundwater, (B) on the concentration and dispersal of pollutants, or their byproducts, through biological, physical, and chemical processes, and (C) on the effects of pollutants on biological community diversity, productivity, and stability, including information on the factors affecting rates of eutrophication and

On November 28, 1980, EPA announced the availability of final ambient water quality criteria documents for 64 of the 65 pollutants designated as toxic under section 307(a)(1) of the Clean Water Act (45 FR 79318). An initial draft criteria document for the remaining pollutant, TCDD, was made available for public comment on March 15, 1979 (44 FR 15928). After consideration of all comments received from the public and peer reviewers, final criteria for TCDD have been derived.

A complete description of the water quality criteria program, including the use of criteria in water quality standards, the methodologies used to derive criteria, and the response to public comment were published in the Federal Register on November 28, 1980 (45 FR 79318).

Summary of Criteria:**2,3,7,8-TETRACHLORODIBENZO-P-DIOXIN****Aquatic Life**

Not enough data are available concerning the effects of 2,3,7,8-TCDD on aquatic life and its uses to allow derivation of national criteria. The available information indicates that acute values for some freshwater animal species are above 1.0 µg/l; some chronic values are below 0.01 µg/l, and the chronic value for rainbow trout is below 0.001 µg/l. Because exposures of some species of fishes to 0.01 µg/l for less than six days resulted in substantial mortality several weeks later, derivation of aquatic life criteria for 2,3,7,8-TCDD may require special consideration. Estimated bioconcentration factors (BCFs) for 2,3,7,8-TCDD range from 3,000 to 900,000, but the available measured BCFs range from 330 to 13,000. If the BCF is 5,000, concentrations above 0.00001 µg/l should result in concentrations in edible freshwater and saltwater fish and shellfish that exceed levels identified in an FDA health advisory. If the BCF is greater than 5,000 or if uptake in a field situation is greater than that in laboratory tests, concentrations of less than 0.00001 µg/l could result in exceedence of levels in the FDA health advisory.

Human Health

For the maximum protection of human health from the potential carcinogenic effects due to exposure to 2,3,7,8-TCDD through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero based on the non-threshold

assumption for this chemical. How zero level may not be attainable at present time. Therefore, the levels may result in an increase of cancer over the lifetime are estimated at 10^{-6} , and 10^{-5} . The corresponding recommended criteria are 1.3×10^{-5} µg/L and 1.3×10^{-6} µg/L respectively. If the above estimates made for consumption of aquatic organisms only, excluding consumption of water, the levels are 1.4×10^{-5} µg/L and 1.4×10^{-6} µg/L respectively. Other concentrations representing different risk levels are calculated. The risk estimate range presented for information purposes does not represent an Agency judgment on an "acceptable" risk level.

Dated: February 2, 1984.

Jack E. Raven,
Assistant Administrator for Office of Water
(FR Doc. 84-3582 Filed 2-15-84; 844 am)
BILLING CODE 6050-00-01

DISTRIBUTION OF THE DISTANCE FROM LAND DISPOSAL UNITS
TO SURFACE WATER BODIES

Prepared by

Environmental Photographic Interpretation Center/ORD
Warrenton, VA.

Prepared for

Office of Solid Wastes
U. S. Environmental Protection Agency
Washington, D.C.

December 11, 1985



UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
ENVIRONMENTAL PHOTOGRAPHIC INTERPRETATION CENTER
VINT HILL FARM STATION
P.O. 1587, WARRENTON, VIRGINIA 22186

December 13, 1985

SUBJECT: RCRA Site Verification Support

FROM: Gordon E. Howard, Jr. *Gordon E. Howard Jr.*
Environmental Scientist
Environmental Photographic Interpretation Center
Advanced Monitoring Systems Division

TO: Edie Pines (WH562-8)
Studies & Methods Branch
Characterization and Assessment Division

The Environmental Photographic Interpretation Center, a field station of the Environmental Monitoring Systems Laboratory, Las Vegas, is providing the Studies and Methods Branch with a RCRA site verification analysis of five classes of permitted features, these are Landfills (D-80); Land Application (D-81); Surface Impoundment, Disposal (D-83); Waste Piles (S-03); and Surface Impoundments, Storage (S-04).

The sites specified are plotted on USGS 7 1/2 minute topographic maps and checked by reviewing the most currently available aerial photography to determine the accuracy of site location. Updated geographic coordinate information is provided for the center of the facility and the permitted feature.

Measurements are made to three classes of water related features (wetlands, perennial streams, and impounded water bodies) if they are located within 1 mile of the site.

The data is provided in tabular form.

Attachments

0	JOB	1113	REF1961	REF2371	REF	ALLOCATED TO SYS00091		
0	JOB	1113	REF1961	REF2051		CATALOG WKCAT02A		KEPT
0	JOB	1113	REF1961	REF2051		VOL SER NOS= UCAT20.		
0	JOB	1113	ICM700011	GGE		LAST ACCESS AT 00:20:54 ON TUESDAY, DECEMBER 17, 1985		
0	JOB	1113	SHASP373	GGE13		STARTED - INIT 13 - CLASS A - SYS EP2A		
0	JOB	1113	MC00051	= JOB GGE13		ENDED 12/17/85 AT 0:53:05, PRTY=02, CC=0000		
0	JOB	1113	SHASP395	GGE13		ENDED		

```

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MCC9501  EXCPS: 105 AT 9.60/1000
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MCC9491  -----
JEF3731  STEP /SAS / START 05351.0052
JEF3741  STEP /SAS / STOP 05351.0052 CPU 0MIN 00.165EC SMO 0MIN 00.025EC VIRT 280K SYS 0032K
JEF3751  GGE13 SAS A0276 - STEP HAS EXECUTED - CODE CODE 0000
MCC9501  ----- U.S.EPA SYSTEM EPA2 - STEP SUMMARY -----
MCC9491  JOB GGE13 STEP SAS CC: 0000
MCC9501  START TUESDAY 12/17/85 AT 0:52:48 STOP 12/17/85 AT 0:53:05
MCC9501  0:17.07 ELAPSED 0:00.50 TCB 0:00.05 SMO 0:00.55 TOT CPU
MCC9491  -----
MCC9501  MEMORY: 0K VIRTUAL ADDRESS SPACE. 14124K
MCC9501  EXCPS: 357 DA, 0 MT, 0 OTHER, 357 TOTAL
MCC9501  EXCPS BY UNIT: 002: 0 000: 0 002: 0
MCC9501  048: 0 050: 0 050: 0 050: 0
MCC9501  058: 0 090: 0 090: 0 090: 0
MCC9501  060: 0 090: 0 090: 0 090: 0
MCC9501  061: 0 077: 0 049: 15 007: 0
MCC9501  000: 0 048: 0 007: 0 007: 0
MCC9501  PAGES IN: 0 VIO 0 SMAP 0 OTHER
MCC9501  PAGES OUT: 0 VIO 0 SMAP 0 OTHER
MCC9491  107 PAGE SECONDS 13.67 RESIDENT SECONDS
MCC9501  CPU : 0:00.55 AT 91050./MR
MCC9501  EXCPS: 357 AT 9.60/1000
MCC9501  TOTAL COST FOR STEP SAS AT PRY=2 (1.0) =
MCC9491  -----
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JEF3741  STEP /SAS / STOP 05351.0053 CPU 0MIN 00.505EC SMO 0MIN 00.055EC VIRT 500K SYS 292K EXT 0032K
MCC9511  ----- U.S.EPA SYSTEM EPA2 - JOB SUMMARY -----
MCC9491  JOB GGE13 0002 STEPS CC: 0000
MCC9511  SUBMIT TUESDAY 12/17/85 AT 0:52:19
MCC9511  START 12/17/85 AT 0:52:37 STOP 12/17/85 AT 0:53:05
MCC9511  0:20.06 ELAPSED 0:00.46 TCB 0:00.07 SMO 0:00.73 TOT CPU
MCC9491  -----
MCC9511  CPU : 0:00.73 AT 91050./MR
MCC9511  EXCPS: 462 AT 9.60/1000
MCC9511  TOTAL COST FOR JOB GGE13 AT PRY=2 (1.0) =
MCC9491  -----
JEF3751  JOB /GGE13 / START 05351.0052
JEF3761  JOB /GGE13 / STOP 05351.0053 CPU 0MIN 00.465EC SMO 0MIN 00.075EC

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99	100

NOTE: COPYRIGHT (C) 1984 SAS INSTITUTE INC., CARY, N.C. 27511, U.S.A.
NOTE: THE JOB 00E13 HAS BEEN RUN UNDER RELEASE 5.06 OF SAS AT THE ENVIRONMENTAL PROTECTION AGENCY (01109001).

NOTE: CPUID VERSION = 20 SERIAL = 170109 MODEL = 3090
CPUID VERSION = 20 SERIAL = 270109 MODEL = 3090

NOTE: SAS OPTIONS SPECIFIED ARE:
MACRO MDOVP SORT=4

```
1
2 TITLE 'DATA REQUEST FOR EYDIE PINES';
3 TITLE 'PREPARED BY OPRA, REQUEST NUMBER A0276';
4 TITLE 'DATA SOURCE: EPIC DATA';
5
6 DATA MASTER;
7 SET EPIC.UPDATE1;
8 CLOSEST = MIN(SYNDIST,LAKEDIST,MAKEDIST);
9
```

NOTE: MISSING VALUES WERE GENERATED AS A RESULT OF PERFORMING
AN OPERATION ON MISSING VALUES.
EACH PLACE IS GIVEN BY: (NUMBER OF TIMES) AT (LINE):(COLUMN).

15 AT 0:13

NOTE: DATA SET WORK.MASTER HAS 346 OBSERVATIONS AND 32 VARIABLES. 210 OBS/TRK.
NOTE: THE DATA STATEMENT USED 0.00 SECONDS AND 540K.

```
10 PROC UNIVARIATE DATA=MASTER;
11 VAR CLOSEST;
12 TITLE4 'DISTANCES TO THE NEAREST SURFACE WATER';
13 TITLES 'DISTANCES ARE IN FEET';
14
```

NOTE: THE PROCEDURE UNIVARIATE USED 0.07 SECONDS AND 760K AND PRINTED PAGE 1.

```
15 DATA ONE;
16 SET MASTER;
17 LENGTH DISTANCE $16;
18 DISTANCE = ' ';
19 IF CLOSEST < 0 THEN DO; DISTANCE='NO SURFACE WATER'; OUTPUT RETURN; END;
20 IF CLOSEST >= 0 AND CLOSEST <= 500 THEN DO; DISTANCE='0000-0500';
21 OUTPUT; RETURN; END;
22 IF CLOSEST > 500 AND CLOSEST <= 1000 THEN DO; DISTANCE='0501-1000';
23 OUTPUT; RETURN; END;
24 IF CLOSEST > 1000 AND CLOSEST <= 1500 THEN DO; DISTANCE='1001-1500';
25 OUTPUT; RETURN; END;
26 IF CLOSEST > 1500 AND CLOSEST <= 2000 THEN DO; DISTANCE='1501-2000';
27 OUTPUT; RETURN; END;
28 IF CLOSEST > 2000 AND CLOSEST <= 2500 THEN DO; DISTANCE='2001-2500';
29 OUTPUT; RETURN; END;
30 IF CLOSEST > 2500 AND CLOSEST <= 3000 THEN DO; DISTANCE='2501-3000';
31 OUTPUT; RETURN; END;
32 IF CLOSEST > 3000 AND CLOSEST <= 3500 THEN DO; DISTANCE='3001-3500';
33 OUTPUT; RETURN; END;
34 IF CLOSEST > 3500 AND CLOSEST <= 4000 THEN DO; DISTANCE='3501-4000';
35 OUTPUT; RETURN; END;
36 IF CLOSEST > 4000 AND CLOSEST <= 4500 THEN DO; DISTANCE='4001-4500';
37 OUTPUT; RETURN; END;
```

8:52 TUESDAY, DECEMBER 17, 1985

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```

2  SASIP1 LOG  OS SAS 5.00      MVS/XA JOB GCE13  STEP A0276  PROC SAS
38  IF CLOSEST > 4500 AND CLOSEST <= 5000 THEN DOI DISTANCE='4501-5000'1
39  OUTPUT RETURN; END;
40  IF CLOSEST > 5000 AND CLOSEST <= 5500 THEN DOI DISTANCE='5001-5500'1
41  OUTPUT RETURN; END;
42  IF CLOSEST > 5500 AND CLOSEST <= 6000 THEN DOI DISTANCE='5501-6000'1
43  OUTPUT RETURN; END;
44  IF CLOSEST > 6000 AND CLOSEST <= 6500 THEN DOI DISTANCE='6001-6500'1
45  OUTPUT RETURN; END;
46

```

NOTE: DATA SET WORK.ONE HAS 346 OBSERVATIONS AND 33 VARIABLES. 200 OBS/TRK.
NOTE: THE DATA STATEMENT USED 0.00 SECONDS AND 544K.

```

47  PROC FREQ;
48  TABLE DISTANCE;
49  LABEL DISTANCE='DISTANCE IN FEET'1
50  TITLE4 'FREQUENCY OF SURFACE WATER DISTANCES AT 500 FT INTERVALS'1
51
52

```

NOTE: THE PROCEDURE FREQ USED 0.07 SECONDS AND 952K AND PRINTED PAGE 2.

```

53  PROC FREQ DATA=MASTER;
54  TABLE CLOSEST;
55  LABEL CLOSEST='DISTANCE IN FEET'1
56  TITLE4 'FREQUENCY OF SURFACE WATER DISTANCES'1
57
58

```

NOTE: THE PROCEDURE FREQ USED 0.11 SECONDS AND 976K AND PRINTED PAGES 3 TO 0.
NOTE: SAS USED 976K MEMORY.

NOTE: SAS INSTITUTE INC.
SAS CIRCLE
PO BOX 8000
CARY, N.C. 27511-0000

8:52 TUESDAY, DECEMBER 17, 1965 1

DATA REQUEST FOR CYDIE PINES
PREPARED BY DPMIA, REQUEST NUMBER A0276
DATA, SOURCE: EPIC DATA
DISTANCES TO THE NEAREST SURFACE WATER
DISTANCES ARE IN FEET

UNIVARIATE

VARIABLE-CLOSEST

MOMENTS

N	351	SUM MGTS	351
MEAN	1060.75	SUM	375131
STD DEV	1235.91	VARIANCE	1527402
SKEWNESS	1.35404	KURTOSIS	1.31337
UNSS	935539757	CSS	534618760
CV	115.641	STD MEAN	65.9602
T-MEAN-0	16.201	PROB> T	0.0001
SIGN RANK	17009	PROB> S	0.0001
NOM - 0	267		

QUANTILES(DEF=4)

100% MAX	6300	99%	4501.64
75% Q3	1696	95%	3740.50
50% MED	637	90%	3055.4
25% Q1	01	10%	0
0% MIN	0	5%	0
		1%	0
RANGE	6300		
Q3-Q1	1615		
MODE	0		

MISSING VALUE	15
COUNT	4.10
% COUNT/NOM	

EXTREMES

LOWEST	0	HIGHEST	4449
	0		4561
	0		4604
	0		5309
	0		6300

DATA REQUEST FOR EYDIE PINES
 PREPARED BY DPMR, REQUEST NUMBER A0276
 DATA SOURCE: EPIC DATA

0:52 TUESDAY, DECEMBER 17, 1985 2

FREQUENCY OF SURFACE WATER DISTANCES AT 500 FT INTERVALS

DISTANCE IN FEET

DISTANCE	FREQUENCY	PERCENT	CUMULATIVE FREQUENCY	CUMULATIVE PERCENT
NO SURFACE WATER	15	4.1	15	4.1
0000-0500	163	44.5	178	48.6
0501-1000	51	13.9	229	62.6
1001-1500	39	10.7	268	73.2
1501-2000	32	8.7	300	82.0
2001-2500	16	4.4	316	86.3
2501-3000	14	3.8	330	90.2
3001-3500	14	3.8	344	94.0
3501-4000	6	1.6	350	95.6
4001-4500	12	3.3	362	98.9
4501-5000	2	0.5	364	99.5
5001-5500	1	0.3	365	99.7
6001-6500	1	0.3	366	100.0

0:52 TUESDAY, DECEMBER 17, 1985 3

DATA REQUEST FOR EYDIE PINES
PREPARED BY DPM, REQUEST NUMBER A0276
DATA SOURCE: EPIC DATA
FREQUENCY OF SURFACE WATER DISTANCES

DISTANCE IN FEET

CLOSEST	FREQUENCY	PERCENT	CUMULATIVE FREQUENCY	CUMULATIVE PERCENT
0	15	23.9	04	23.9
50	1	0.3	05	24.2
72	1	0.3	06	24.5
77	1	0.3	07	24.8
81	1	0.3	08	25.1
83	1	0.3	09	25.4
84	1	0.3	10	25.6
88	1	0.3	11	25.9
89	1	0.3	12	26.2
94	1	0.3	13	26.5
96	1	0.3	14	26.8
105	1	0.3	15	27.1
111	2	0.6	17	27.6
119	1	0.3	18	27.9
122	1	0.3	19	28.2
125	1	0.3	20	28.5
133	1	0.3	21	28.8
140	3	0.9	24	29.6
144	1	0.3	25	29.9
145	1	0.3	26	30.2
154	1	0.3	27	30.5
159	1	0.3	28	30.8
161	1	0.3	29	31.1
163	1	0.3	30	31.3
170	1	0.3	31	31.6
171	1	0.3	32	31.9
181	1	0.3	33	32.2
183	1	0.3	34	32.5
186	1	0.3	35	32.8
188	2	0.6	37	33.3
190	1	0.3	38	33.6
192	1	0.3	39	33.9
195	1	0.3	40	34.2
204	1	0.3	41	34.5
210	1	0.3	42	34.8
219	2	0.6	44	35.3
222	1	0.3	45	35.6
231	1	0.3	46	35.9
234	1	0.3	47	36.2
245	2	0.6	49	36.6
251	1	0.3	50	37.0
253	1	0.3	51	37.3
255	1	0.3	52	37.6
257	1	0.3	53	37.9
265	1	0.3	54	38.2
267	1	0.3	55	38.5
269	1	0.3	56	38.7
273	1	0.3	57	39.0

691

0:52 TUESDAY, DECEMBER 17, 1965 4

DATA REQUEST FOR EYDIE PIMES
PREPARED BY DPMIA, REQUEST NUMBER 40276
DATA SOURCE: EPIC DATA
FREQUENCY OF SURFACE WATER DISTANCES

DISTANCE IN FEET

CLOSEST	FREQUENCY	PERCENT	CUMULATIVE FREQUENCY	CUMULATIVE PERCENT
283	1	0.3	141	40.2
301	1	0.3	142	40.5
307	1	0.3	143	40.7
319	1	0.3	144	41.0
336	1	0.3	145	41.3
350	1	0.3	146	41.6
360	1	0.3	147	41.9
363	1	0.3	148	42.2
375	1	0.3	149	42.5
379	2	0.6	151	43.0
387	1	0.3	152	43.3
406	1	0.3	153	43.6
426	1	0.3	154	43.9
431	1	0.3	155	44.2
437	1	0.3	156	44.4
443	1	0.3	157	44.7
457	1	0.3	158	45.0
462	1	0.3	159	45.3
465	2	0.6	161	45.9
474	1	0.3	162	46.2
478	1	0.3	163	46.4
502	1	0.3	164	46.7
513	1	0.3	165	47.0
516	1	0.3	166	47.3
537	1	0.3	167	47.6
544	1	0.3	168	47.9
558	1	0.3	169	48.1
569	1	0.3	170	48.4
590	1	0.3	171	48.7
599	1	0.3	172	49.0
606	1	0.3	173	49.3
615	1	0.3	174	49.6
616	1	0.3	175	49.9
637	2	0.6	177	50.4
643	1	0.3	178	50.7
651	1	0.3	179	51.0
653	1	0.3	180	51.3
666	1	0.3	181	51.6
674	2	0.6	183	52.1
688	1	0.3	184	52.4
689	1	0.3	185	52.7
733	1	0.3	186	53.0
734	1	0.3	187	53.3
764	1	0.3	188	53.6
776	1	0.3	189	53.9
777	1	0.3	190	54.1
788	1	0.3	191	54.4
784	1	0.3	192	54.7
789	1	0.3	193	55.0
815	1	0.3	194	55.3

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DATA REQUEST FOR EYDIE PINES
PREPARED BY DFWA, REQUEST NUMBER A0276
DATA SOURCE: EPIC DATA
FREQUENCY OF SURFACE WATER DISTANCES

DISTANCE IN FEET				
CLOSEST	FREQUENCY	PERCENT	CUMULATIVE FREQUENCY	CUMULATIVE PERCENT
820	1	0.3	195	55.6
851	1	0.3	196	55.8
852	1	0.3	197	56.1
865	1	0.3	198	56.4
883	2	0.6	200	57.0
885	1	0.3	201	57.3
886	1	0.3	202	57.5
896	1	0.3	203	57.8
916	1	0.3	204	58.1
921	1	0.3	205	58.4
930	1	0.3	206	58.7
937	1	0.3	207	59.0
976	1	0.3	208	59.3
977	2	0.6	210	59.8
978	1	0.3	211	60.1
990	1	0.3	212	60.4
991	1	0.3	213	60.7
993	1	0.3	214	61.0
1009	1	0.3	215	61.3
1012	1	0.3	216	61.5
1036	1	0.3	217	61.8
1037	1	0.3	218	62.1
1049	1	0.3	219	62.4
1051	1	0.3	220	62.7
1059	1	0.3	221	63.0
1062	1	0.3	222	63.2
1068	1	0.3	223	63.5
1082	1	0.3	224	63.8
1104	1	0.3	225	64.1
1115	1	0.3	226	64.4
1126	1	0.3	227	64.7
1129	1	0.3	228	65.0
1133	1	0.3	229	65.2
1137	1	0.3	230	65.5
1151	1	0.3	231	65.8
1154	1	0.3	232	66.1
1177	1	0.3	233	66.4
1223	1	0.3	234	66.7
1225	1	0.3	235	67.0
1239	1	0.3	236	67.2
1249	1	0.3	237	67.5
1256	1	0.3	238	67.8
1264	1	0.3	239	68.1
1268	1	0.3	240	68.4
1291	1	0.3	241	68.7
1306	1	0.3	242	68.9
1310	1	0.3	243	69.2
1362	1	0.3	244	69.5
1378	1	0.3	245	69.8

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DATA REQUEST FOR EYDIE PINES
 PREPARED BY DPRA, REQUEST NUMBER A0276
 DATA SOURCE: EPIC DATA
 FREQUENCY OF SURFACE WATER DISTANCES

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1 DISTANCE IN FEET

CLOSEST	FREQUENCY	PERCENT	CUMULATIVE FREQUENCY	CUMULATIVE PERCENT
1305	1	0.3	247	70.4
1390	1	0.3	248	70.7
1415	1	0.3	249	70.9
1449	1	0.3	250	71.2
1450	1	0.3	251	71.5
1475	1	0.3	252	71.8
1491	1	0.3	253	72.1
1517	1	0.3	254	72.4
1518	1	0.3	255	72.6
1533	1	0.3	256	72.9
1545	1	0.3	257	73.2
1550	1	0.3	258	73.5
1572	1	0.3	259	73.8
1612	1	0.3	260	74.1
1647	1	0.3	261	74.4
1657	1	0.3	262	74.6
1681	1	0.3	263	74.9
1696	1	0.3	264	75.2
1700	1	0.3	265	75.5
1710	1	0.3	266	75.8
1729	1	0.3	267	76.1
1731	1	0.3	268	76.4
1741	1	0.3	269	76.6
1753	1	0.3	270	76.9
1760	1	0.3	271	77.2
1782	1	0.3	272	77.5
1816	1	0.3	273	77.8
1819	1	0.3	274	78.1
1826	1	0.3	275	78.3
1852	1	0.3	276	78.6
1856	1	0.3	277	78.9
1868	1	0.3	278	79.2
1871	1	0.3	279	79.5
1910	1	0.3	280	79.8
1916	1	0.3	281	80.1
1922	1	0.3	282	80.3
1937	1	0.3	283	80.6
1941	1	0.3	284	80.9
1980	1	0.3	285	81.2
2057	1	0.3	286	81.5
2134	1	0.3	287	81.8
2139	1	0.3	288	82.1
2140	1	0.3	289	82.3
2159	1	0.3	290	82.6
2174	1	0.3	291	82.9
2236	1	0.3	292	83.2
2305	1	0.3	293	83.5
2307	1	0.3	294	83.8
2410	1	0.3	295	84.1
2416	1	0.3	296	84.4

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DATA REQUEST FOR EYOIE PINES
 PREPARED BY DPMR, REQUEST NUMBER A0276
 DATA SOURCE: EPIC DATA
 FREQUENCY OF SURFACE WATER DISTANCES

DISTANCE IN FEET

CLOSEST	FREQUENCY	PERCENT	CUMULATIVE FREQUENCY	CUMULATIVE PERCENT
2437	1	0.3	297	84.6
2450	1	0.3	298	84.9
2456	1	0.3	299	85.2
2476	1	0.3	300	85.5
2500	1	0.3	301	85.8
2512	1	0.3	302	86.0
2602	1	0.3	303	86.3
2610	1	0.3	304	86.6
2623	1	0.3	305	86.9
2690	1	0.3	306	87.2
2693	1	0.3	307	87.5
2750	1	0.3	308	87.7
2804	1	0.3	309	88.0
2835	1	0.3	310	88.3
2852	1	0.3	311	88.6
2902	1	0.3	312	88.9
2966	1	0.3	313	89.2
2967	1	0.3	314	89.5
2981	1	0.3	315	89.7
3041	1	0.3	316	90.0
3059	1	0.3	317	90.3
3065	1	0.3	318	90.6
3081	1	0.3	319	90.9
3114	1	0.3	320	91.2
3144	1	0.3	321	91.5
3161	1	0.3	322	91.7
3263	1	0.3	323	92.0
3286	1	0.3	324	92.3
3325	1	0.3	325	92.6
3369	1	0.3	326	92.9
3401	1	0.3	327	93.2
3415	1	0.3	328	93.4
3490	1	0.3	329	93.7
3505	1	0.3	330	94.0
3509	1	0.3	331	94.3
3539	1	0.3	332	94.6
3592	1	0.3	333	94.9
3659	1	0.3	334	95.2
3663	1	0.3	335	95.4
4071	1	0.3	336	95.7
4087	1	0.3	337	96.0
4090	1	0.3	338	96.3
4132	1	0.3	339	96.6
4196	1	0.3	340	96.9
4199	1	0.3	341	97.2
4204	1	0.3	342	97.4
4223	1	0.3	343	97.7
4254	1	0.3	344	98.0
4321	1	0.3	345	98.3

DATA REQUEST FOR EYDIE PINES
 PREPARED BY BFWA. REQUEST NUMBER A0276
 DATA SOURCE: EPIC DATA
 FREQUENCY OF SURFACE WATER DISTANCES

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DISTANCE IN FEET

CLOSEST	FREQUENCY	PERCENT	CUMULATIVE FREQUENCY	CUMULATIVE PERCENT
4449	1	0.3	347	90.9
4561	1	0.3	348	91.1
4404	1	0.3	349	91.4
5389	1	0.3	350	91.7
6300	1	0.3	351	100.0

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ANALYTIC SOLUTION TO THE TRANSIENT AND STEADY-STATE
MASS TRANSPORT EQUATION MODELLING 2-D GROUND WATER FLOW
WITH A GAUSSIAN SOURCE IN A FINITE WIDTH AQUIFIER

Prepared by

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3746 Mt Diablo Blvd
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Prepared for

Office of Solid Waste
U. S. Environmental Protection Agency
Washington, D.C.

Analytic Solution To The Transient
 And Steady-State Mass Transport Equation
 Modeling 2-D Ground Water Flow
 With A Gaussian Source (1st type b.c.)
 In A Finite Width Aquifer

$$0 \leq x < \infty$$

semi-infinite length

$$0 \leq y \leq W$$

finite width

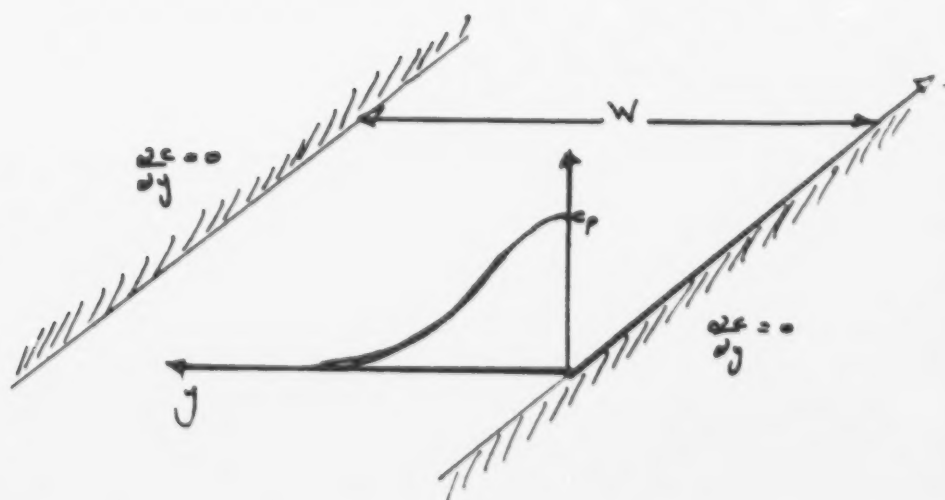
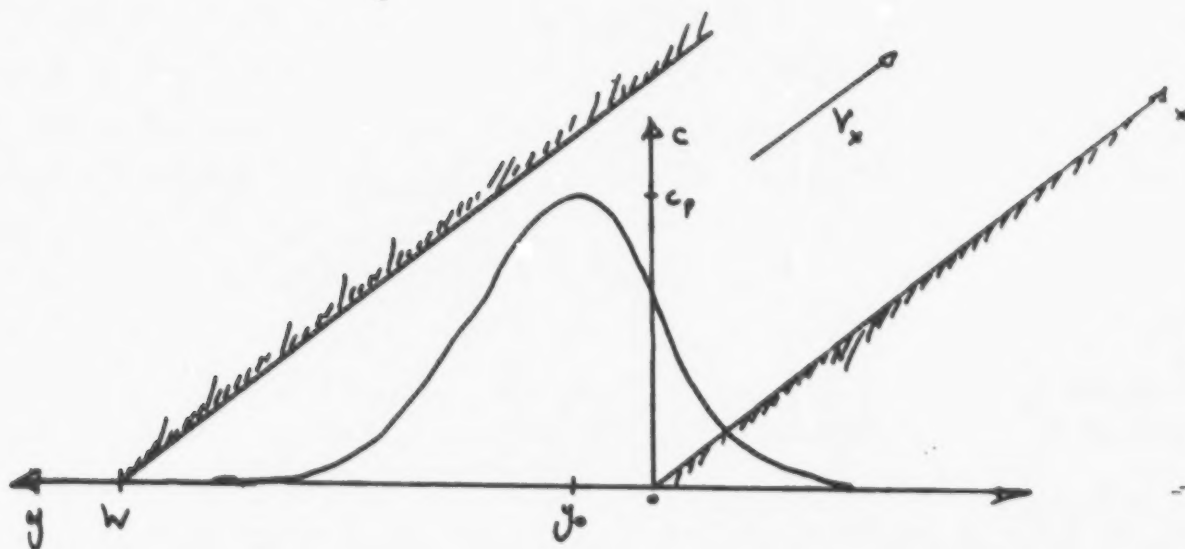
considers

velocity: V_x

dispersions: D_x, D_y

Michael J. Uryu
 September 2, 1985

Gaussian Source



Source

$$c(0, y, t) = c_p e^{-\frac{(y-y_0)^2}{2S^2}} \cdot \left\{ U(t-t_{on}) - U(t-t_{off}) \right\}$$

c_p maximum source concentration $[M_{solub}/L^3]$

$$c(0, y_0, t) = c_p$$

S standard deviation $[L]$

$$S = \frac{1}{c_p \sqrt{2\pi}} \int_{y'=-\infty}^{\infty} c(0, y', t) dy' \quad [L]$$

The contaminant mass flux in $[M/T]$ across the section $x=0$ is given by:

$$\dot{m} = \theta V_x \Delta z \int_{y=0}^W c_p e^{-\frac{(y-y_0)^2}{2S^2}} dy \quad [M/T]$$

$$\dot{m} = \theta V_x \Delta z c_p \frac{S\sqrt{2\pi}}{2} \left\{ \operatorname{erf}\left(\frac{W-y_0}{S\sqrt{2}}\right) + \operatorname{erf}\left(\frac{y_0}{S\sqrt{2}}\right) \right\}$$

If $\dot{m}$ is given, solve for S

Notation

- x, y spatial location $[L]$
 c liquid phase solute concentration $[M_{\text{solute}}/L^3_w]$
 c_p maximum or peak solute concentration $[M_{\text{solute}}/L^3_w]$
 $\rightarrow$ solid phase solute concentration $[M_{\text{solute}}/M_{\text{solid}}]$
 $s = k_d c$
 k_d partition or distribution coefficient for linear, equilibrium sorption $[L^3_w/M_{\text{solid}}]$
 k first-order decay rate $[T^{-1}]$
 D_x, D_y dispersion coefficients $[L^2/T]$
 V_x average or pore-water velocity $[L/T]$
 R_d retardation coefficient for linear, equilibrium sorption $[1]$
 $R_d = 1 + k_d \rho_b$
 θ volumetric moisture content $[L^3_w/L^3]$
 ρ_b soil bulk density $[M_{\text{solid}}/L^3]$
 W width of aquifer $[L]$
 y_0 center of Gaussian source $[L]$
 t time $[T]$

$$R_d = \frac{(k_{\text{liquid}} + k_{\text{solid}} k_d \rho_b)}{k_d}$$

The 2-D mass transport equation is defined as follows

$$1) \quad R_d \frac{\partial c}{\partial t} = D_x \frac{\partial^2 c}{\partial x^2} + D_y \frac{\partial^2 c}{\partial y^2} - V_x \frac{\partial c}{\partial x} - k c R_d$$

with the following initial condition

$$2) \quad c(x, y, 0) = 0$$

and boundary conditions

$$3) \quad \left\{ \begin{array}{l} c(0, y, t) = c_p e^{-\frac{(y-y_0)^2}{2\sigma^2}} \cdot \left\{ U(t-t_{0n}) - U(t-t_{0n}+\tau) \right\} \\ c(\infty, y, t), \frac{\partial c}{\partial x}(\infty, y, t) = 0 \\ \frac{\partial c}{\partial y}(x, 0, t) = 0 \\ \frac{\partial c}{\partial y}(x, w, t) = 0 \end{array} \right. \quad 0 \leq y_0 \leq w$$

Introduce the following transform to reduce Eq. 1) to that of the diffusion equation:

$$1) \quad c(x, y, t) = \hat{c}(x, y, t) \exp \left\{ \frac{V_x x}{2D_x} - \left(\frac{V_x^2}{4D_x R_d} + k \right) t \right\}$$

Take the derivatives of Eq. 1)

$$2) \quad \left\{ \begin{array}{l} \frac{\partial c}{\partial x} = \frac{\partial \hat{c}}{\partial x} e^{\square} + \frac{V_x}{2D_x} \hat{c} e^{\square} \\ \frac{\partial^2 c}{\partial x^2} = \frac{\partial^2 \hat{c}}{\partial x^2} e^{\square} + \frac{V_x}{D_x} \frac{\partial \hat{c}}{\partial x} e^{\square} + \frac{V_x^2}{4D_x^2} \hat{c} e^{\square} \\ \frac{\partial c}{\partial y} = \frac{\partial \hat{c}}{\partial y} e^{\square} \\ \frac{\partial^2 c}{\partial y^2} = \frac{\partial^2 \hat{c}}{\partial y^2} e^{\square} \\ \frac{\partial c}{\partial t} = - \left(\frac{V_x^2}{4D_x R_d} + k \right) \hat{c} e^{\square} + \frac{\partial \hat{c}}{\partial t} e^{\square} \end{array} \right.$$

where

$$6) \quad \square = \left\{ \frac{V_x x}{2D_x} - \left(\frac{V_x^2}{4D_x R_d} + k \right) t \right\}$$

Substitute Eqs 4-6) into Eqs. 1-3) and rearrange to get:

$$7) \quad R_d \frac{\partial \hat{c}}{\partial t} = D_x \frac{\partial^2 \hat{c}}{\partial x^2} + D_y \frac{\partial^2 \hat{c}}{\partial y^2}$$

subject to the initial condition

$$8) \quad \hat{c}(x, y, 0) = 0$$

and boundary conditions

$$9) = \begin{cases} \hat{c}(0, y, t) = c_p a_{ys} \left\{ -\frac{(y-y_0)^2}{2S^2} + \left(\frac{V_x^2}{4D_x R_d} + k \right) t \right\} \cdot \left\{ U(t-t_{0u}) - U(t-t_{0fu}) \right\} & \text{at } x=0 \\ \hat{c}, \frac{\partial \hat{c}}{\partial x} = 0 & \text{at } x = \infty \\ \frac{\partial \hat{c}}{\partial y}(x, 0, t) = \frac{\partial \hat{c}}{\partial y}(x, W, t) = 0 \end{cases}$$

Consider the following integral transform for the semi-infinite x region with a specified first type boundary condition at $x=0$:

$$1) \quad \bar{E}(p, y, t) = \int_{x'=0}^{\infty} \sqrt{\frac{x}{p}} \sin(px) E(x', y, t) dx'$$

and its inverse

$$2) \quad E(x, y, t) = \int_{p=0}^{\infty} \sqrt{\frac{x}{p}} \sin(px) \bar{E}(p, y, t) dp$$

The integral transform for the finite y region with two second type boundary conditions is:

$$1) \quad \bar{E}(p, v_n, t) = \int_{y'=0}^W A_n \frac{\cos(v_n y')}{\sqrt{W}} \bar{E}(p, y', t) dy'$$

and its inverse

$$2) \quad \bar{E}(p, y, t) = \sum_{n=0}^{\infty} \frac{A_n \cos(v_n y)}{\sqrt{W}} \bar{E}(p, v_n, t)$$

where $A_n = \begin{cases} 1 & \text{if } n=0 \\ \sqrt{2} & \text{if } n \neq 0 \end{cases} ; v_n = \frac{n\pi}{W}$

The x variation in Eqn 2-9) can be removed by multiplying these equations by $\sqrt{\frac{2}{\pi}} \sin \beta x'$ and then integrating wrt x' from $0 \rightarrow \infty$:

$$1) R_d \frac{\partial \bar{\epsilon}}{\partial t} = \left\{ D_x \sqrt{\frac{2}{\pi}} \sin \beta x' \bar{\epsilon} \right\} \Big|_{x'=0}^{\infty} - \left\{ \beta D_x \sqrt{\frac{2}{\pi}} \cos \beta x' \bar{\epsilon} \right\} \Big|_{x'=0}^{\infty} - \beta^2 D_x \bar{\epsilon} + D_y \frac{\partial^2 \bar{\epsilon}}{\partial y^2}$$

From substitution of the boundary condition of Eq. 9) into Eq. 14), then

$$1) R_d \frac{\partial \bar{\epsilon}}{\partial t} = -\beta^2 D_x \bar{\epsilon} + D_y \frac{\partial^2 \bar{\epsilon}}{\partial y^2} + c_p \beta D_x \sqrt{\frac{2}{\pi}} e^{\left\{ -\frac{(y-y_0)^2}{2s^2} + \left(\frac{V_x^2}{4D_x R_d} + k \right) t \right\}}$$

The y variation in Eq. 15) can be removed by multiplying it by $A_n \frac{\cos(v_n y')}{\sqrt{W}}$ and then integrating the resultant equation wrt y' from $0 \rightarrow W$, using the relation of Eq. 12) to get:

$$1) R_d \frac{d \bar{\epsilon}}{dt} = -\beta^2 D_x \bar{\epsilon} - v_n^2 D_y \bar{\epsilon} + c_p \beta D_x \sqrt{\frac{2}{\pi}} \frac{A_n}{\sqrt{W}} e^{\left(\frac{V_x^2}{4D_x R_d} + k \right) t} \int_{y'=0}^W \cos(v_n y') e^{-\frac{(y'-y_0)^2}{2s^2}} dy'$$

$$+ \left\{ D_y \frac{A_n}{\sqrt{W}} \cos(v_n y') \frac{d \bar{\epsilon}}{dy'} \right\} \Big|_{y'=0}^W + \left\{ D_y \frac{A_n}{\sqrt{W}} v_n \sin(v_n y') \bar{\epsilon} \right\} \Big|_{y'=0}^W$$

where $v_n = \frac{n\pi}{W}$ $n=0, 1, \dots$

Eg. 6) reduces to the following:

$$17) \quad \frac{d\bar{c}}{dt} = - \left\{ \frac{\beta^2 D_x + v_N^2 D_y}{R_d} \right\} \bar{c} + c_p \frac{\beta D_x}{R_d} \sqrt{\frac{z}{\pi}} \frac{A_N}{\sqrt{W}} e^{\left(\frac{v_N^2}{4 D_x R_d} + k \right) t} \cdot \int_{y'=0}^w \cos(v_N y') e^{-\frac{(y'-y_0)^2}{2 S^2}} dy'$$

Eg. 17) is an ODE wrt t and can be integrated using the initial condition of Eg. 8):

$$8) \quad \bar{c}(p, v_N, t) = c_p \frac{\beta D_x}{R_d} \sqrt{\frac{z}{\pi}} \frac{A_N}{\sqrt{W}} e^{\left(\frac{v_N^2}{4 D_x R_d} + k \right) t} \cdot \int_{t'=t_{on}}^{t_{off}} e^{- \left\{ \frac{\beta^2 D_x + v_N^2 D_y}{R_d} + \frac{v_N^2}{4 D_x R_d} + k \right\} (t-t')} \cdot \int_{y'=0}^w \cos(v_N y') e^{-\frac{(y'-y_0)^2}{2 S^2}} dt' dy'$$

Substitute in the following new variable, τ , where

$$9) \quad \begin{cases} \tau = t - t' \\ d\tau = -dt' \end{cases}$$

t'	τ
t_{on}	$\max(0, t - t_{on})$
t_{off}	$\max(0, t - t_{off})$

Also apply the inverse transforms by multiplying Eq.

10) by the following $\sqrt{\frac{z}{\pi}} \sin px$ and integrating wrt p from $0 \rightarrow \infty$,
 multiply by $\sum_{n=0}^{\infty} A_n \frac{\cos(v_n y)}{\sqrt{W}}$ and finally multiply by $e^{\left\{ \frac{v_n x}{2D_x} - \left(\frac{v_n^2}{4D_x R_d} + k \right) z \right\}}$

$$c) \quad c(x, y, z) = \frac{z c_p D_x}{\pi \rho W} e^{\frac{v_n x}{2D_x}} \sum_{n=0}^{\infty} A_n^2 \cos(v_n y) \cdot e^{\max(0, t-t_{off}) \left\{ \frac{v_n^2 D_x}{R_d} + \frac{v_n^2}{4D_x R_d} + k \right\} z}$$

$$\int_{y'=0}^y \cos(v_n y') e^{-\frac{(y-y')^2}{2S^2}} \cdot \int_{p=0}^{\infty} p \sin(px) e^{-\frac{p^2 D_x z}{R_d}} dp dy' dz$$

$z = \max(0, t - t_{off})$

The last RHS integral can be integrated to give:

$$1) \int_{p=0}^{\infty} p \sin(px) e^{-\frac{p^2 D_x z}{R_d}} dp = \frac{x \sqrt{\pi}}{4} \left(\frac{R_d}{D_x} \right)^{1/2} z^{-3/2} e^{-\frac{x^2 R_d}{4 D_x z}}$$

Substitute Eq. 2) into Eq. 20) to get:

$$22) \quad c(x, y, t) = \frac{c_p x}{2W} \sqrt{\frac{R_1}{2D_x}} e^{\frac{V_x x}{2D_x}} \cdot \sum_{n=0}^{\infty} A_n^2 \cos(v_n y) \cdot \int_{y'=0}^W \cos(v_n y) e^{-\frac{(y'-y)^2}{2S^2}} dy'$$

$$\int_{z=\max(0, t-t_{off})}^{\max(0, t-t_{off})} \frac{e^{-\left\{ \frac{V_x^2 D_1}{R_1} + \frac{V_x^2}{4D_x R_1} + h \right\} z} - \frac{x^2 R_1}{4D_x z}}{z^{3/2}} dz$$

$z = \max(0, t - t_{off})$

Note the following integral

$$23) \quad \int e^{-xz} e^{-az - b/z} dz = \frac{1}{2\sqrt{ab}} \left\{ e^{-2\sqrt{ab}} \operatorname{erf}\left(-\sqrt{\frac{b}{a}} + \sqrt{at}\right) - e^{\frac{2\sqrt{ab}}{t}} \operatorname{erf}\left(\sqrt{\frac{b}{a}} + \sqrt{at}\right) \right\}$$

Substitute Eq. 23) into Eq. 22), observing the limits of integration shown in Eq. 22)

$$4) \ c(x, y, t) = \frac{c_p}{2W} e^{\frac{V_x x}{2D_x}} \cdot \sum_{n=0}^{\infty} A_n^2 \cos(v_n y) \cdot \int_{y'=0}^W \cos(v_n y') e^{-\frac{(y'-y_0)^2}{2S_z^2}} dy'.$$

$$\left\{ e^{-2\sqrt{a_n}b} \left[\operatorname{erfc} \left(\sqrt{\frac{b}{m_{xy}(0, t-t_{0n})}} - \sqrt{a_n m_{xy}(0, t-t_{0n})} \right) - \operatorname{erfc} \left(\sqrt{\frac{b}{m_{xy}(0, t-t_{0n})}} - \sqrt{a_n m_{xy}(0, t-t_{0n})} \right) \right] + e^{2\sqrt{a_n}b} \left[\operatorname{erfc} \left(\sqrt{\frac{b}{m_{xy}(0, t-t_{0n})}} + \sqrt{a_n m_{xy}(0, t-t_{0n})} \right) - \operatorname{erfc} \left(\sqrt{\frac{b}{m_{xy}(0, t-t_{0n})}} + \sqrt{a_n m_{xy}(0, t-t_{0n})} \right) \right] \right\}$$

where

$$a_n = \left\{ \frac{v_n^2 D_y}{R_y} + \frac{V_x^2}{4D_x R_y} + R_y \right\}$$

$$b = \left\{ \frac{x^2 R_y}{4D_x} \right\}$$

$$v_n = \frac{n\pi}{W} \quad n=0, 1, 2, \dots$$

$$A_n = \begin{cases} 1 & \text{if } n=0 \\ \sqrt{2} & \text{if } n \neq 0 \end{cases}$$

Special case

let $y_0 = 0$

If $y_0 = 0$, then the integral of Eq. 24) can be evaluated to give:

$$25) \int_{y'=0}^W \exp(v_0 y') e^{-\frac{y'^2}{2s^2}} dy' = \frac{s\sqrt{2\pi}}{4} e^{-\frac{v_0^2 s^2}{2}} \left\{ \operatorname{erf}\left(\frac{W}{s\sqrt{2}} - \frac{iv_0 s}{\sqrt{2}}\right) + \operatorname{erf}\left(\frac{W}{s\sqrt{2}} + \frac{iv_0 s}{\sqrt{2}}\right) \right\}$$

Substitute Eq. 25) into Eq. 24) and rearrange terms to get the final solution

$$\begin{aligned}
 c(x, y, t) &= \frac{Sc_p}{W\sqrt{\frac{\pi}{c}}} e^{\frac{V_x x}{cD_x}} \sum_{n=0}^{\infty} A_n^2 \cos(v_n y) \cdot e^{-\frac{V_x^2 S^2}{c}} \\
 &\cdot \left\{ \operatorname{erf}\left(\frac{W}{S\sqrt{c}} - \frac{iV_x S}{J\sqrt{c}}\right) + \operatorname{erf}\left(\frac{W}{S\sqrt{c}} + \frac{iV_x S}{J\sqrt{c}}\right) \right\} \\
 &\cdot \left\{ e^{-2\sqrt{a_n}b} \left[\operatorname{erfc}\left(\sqrt{\frac{b}{\operatorname{may}(0, t-t_{0n})}} - \sqrt{a_n \operatorname{may}(0, t-t_{0n})}\right) \right. \right. \\
 &\quad \left. \left. - \operatorname{erfc}\left(\sqrt{\frac{b}{\operatorname{may}(0, t-t_{0n})}} + \sqrt{a_n \operatorname{may}(0, t-t_{0n})}\right) \right] \right. \\
 &\quad \left. + e^{-2\sqrt{a_n}b} \left[\operatorname{erfc}\left(\sqrt{\frac{b}{\operatorname{may}(0, t-t_{0n})}} + \sqrt{a_n \operatorname{may}(0, t-t_{0n})}\right) \right. \right. \\
 &\quad \left. \left. - \operatorname{erfc}\left(\sqrt{\frac{b}{\operatorname{may}(0, t-t_{0n})}} - \sqrt{a_n \operatorname{may}(0, t-t_{0n})}\right) \right] \right\}
 \end{aligned}$$

if $0 \dots$

where

$$a_n = \left\{ \frac{V_x^2 D_y}{R_d} + \frac{V_x^2}{4D_x R_d} + R \right\} \quad n=0, 1, 2, \dots$$

$$b = \left\{ \frac{x^2 R_d}{4D_x} \right\}$$

$$v_n = \frac{n\pi}{W} \quad n=0, 1, 2, \dots \quad ; \quad A_n = \begin{cases} 1 & n=0 \\ \sqrt{2} & n \neq 0 \end{cases}$$

Study - State

y.oo

Steady state solution

$t \rightarrow \infty$

$$2P) c(x, y, \infty) = \frac{Sc_p}{WZ} \sqrt{\frac{\pi}{2}} e^{\frac{V_n x}{2D_n}} \sum_{n=0}^{\infty} A_n^2 \cos(v_n y) e^{-\frac{v_n^2 S^2}{2} - 2\sqrt{a_n} b}$$

$$\cdot \left\{ \operatorname{erf}\left(\frac{W}{S\sqrt{2}} - \frac{iv_n S}{\sqrt{2}}\right) + \operatorname{erf}\left(\frac{W}{S\sqrt{2}} + \frac{iv_n S}{\sqrt{2}}\right) \right\}$$

$y=0$

$$v_n = \frac{n\pi}{W} \quad n=0, 1, \dots$$

$$a_n = \left\{ \frac{v_n^2 D_n}{R_d} + \frac{V_n^2}{4D_n R_d} + k \right\} \quad n=0, 1, \dots$$

$$b = \left\{ \frac{x^2 R_d}{4D_n} \right\}$$

$$A_n = \begin{cases} 1 & n=0 \\ \sqrt{2} & n=1, 2, \dots \end{cases}$$